

APPENDIX 2

**SUMMARY OF HUMAN HEALTH RISK-BASED
EQUATIONS AND DEFAULT INPUT PARAMETER
VALUES; USEPA REGION IX PRG DOCUMENT
(OCTOBER 2002, TEXT ONLY)**

INTERIM FINAL – JULY 2003
SF Bay RWQCB

EQUATIONS FOR DERIVATION OF RISK-BASED SCREENING LEVELS FOR SOIL (DIRECT EXPOSURE), INDOOR AIR AND DRINKING WATER

1.0 Introduction

A summary of models and assumptions used to develop for human health, direct-exposure concerns is presented below. For additional information on the models refer to the document *Region IX Preliminary Remediation Goals* ("PRGs", USEPA 2002) and other documents as referenced. A copy of the text of this document is attached.

2.0 SOIL

2.1 Shallow Soils

Human exposure assumptions are summarized in Table 1. With the exception of the construction/trench worker exposure scenario, parameter values in Table 1 were taken directly from the USEPA Region IX PRG document. Parameter values for the construction/trench worker exposure scenario are discussed in more detail in Appendix 1. Tables 2 and 3 summarize equations and parameter values used to develop the PRG Volatilization Factors and Particulate Emission Factor.

Age-Adjusted Exposure Factors

Carcinogenic risks under residential exposure scenarios were calculated using the following age-adjusted factors:

1) ingestion [(mg-yr)/kg-day]]:

$$IFS_{adj} = \frac{ED_c \times IRS_c}{BW_c} + \frac{(ED_r - ED_c) \times IRS_a}{BW_a}$$

2) dermal contact [(mg-yr)/kg-day]]:

$$SFS_{adj} = \frac{ED_c \times AF_c \times SA_c}{BW_c} + \frac{(ED_r - ED_c) \times AF_a \times IRS_a}{BW_a}$$

3) inhalation [(m3-yr)/kg-day]]:

$$InhF_{adj} = \frac{ED_c \times IRA_c}{BW_c} + \frac{(ED_r - ED_c) \times IRA_a}{BW_a}$$

Definition of terms and default parameter values used in the equations are presented in Tables a through c.

Direct exposure equations for soil are summarized as follows:

Equation 1: Combined Exposures to Carcinogenic Contaminants in Residential Soil

$$C(\text{mg/kg}) = \frac{\text{TR} \times \text{AT}_c}{\text{EF}_r \left[\left(\frac{\text{IFS}_{\text{adj}} \times \text{CSF}_o}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{SF}_{\text{adj}} \times \text{ABS} \times \text{CSF}_o}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{InhF}_{\text{adj}} \times \text{CSF}_i}{\text{VF}} \right) \right]}$$

Equation 2: Combined Exposures to Noncarcinogenic Contaminants in Residential Soil

$$C(\text{mg/kg}) = \frac{\text{THQ} \times \text{BW}_c \times \text{AT}_n}{\text{EF}_r \times \text{ED}_c \left[\left(\frac{\text{I}}{\text{RfD}_o} \times \frac{\text{IRS}_c}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{I}}{\text{RfD}_o} \times \frac{\text{SA}_c \times \text{AF}_c \times \text{ABS}}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{I}}{\text{RfD}_i} \times \frac{\text{IRA}_c}{\text{VF}} \right) \right]}$$

Equation 3: Combined Exposures to Carcinogenic Contaminants in Industrial Soil

$$C(\text{mg/kg}) = \frac{\text{TR} \times \text{BW}_a \times \text{AT}_c}{\text{EF}_o \times \text{ED}_o \left[\left(\frac{\text{IRS}_o \times \text{CSF}_o}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{SA}_a \times \text{AF}_a \times \text{ABS} \times \text{CSF}_o}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{IRA}_a \times \text{CSF}_i}{\text{VF}} \right) \right]}$$

Equation 4: Combined Exposures to Noncarcinogenic Contaminants in Industrial Soil

$$C(\text{mg/kg}) = \frac{\text{THQ} \times \text{BW}_a \times \text{AT}_n}{\text{EF}_o \times \text{ED}_o \left[\left(\frac{\text{I}}{\text{RfD}_o} \times \frac{\text{IRS}_o}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{I}}{\text{RfD}_o} \times \frac{\text{SA}_a \times \text{AF}_a \times \text{ABS}}{10^6 \text{mg/kg}} \right) + \left(\frac{\text{I}}{\text{RfD}_i} \times \frac{\text{IRA}_a}{\text{VF}} \right) \right]}$$

Equation 5: Derivation of the Volatilization Factor

$$\text{VF}(\text{m}^3/\text{kg}) = (Q/C) \times \frac{(3.14 \times D_A \times T)^{1/2}}{(2 \times P_b \times D_A)} \times 10^{-4}(\text{m}^2/\text{cm}^2)$$

$$D_A = \frac{[(\Theta_a^{10/3} D_i H^i + \Theta_w^{10/3} D_w)/n^2]}{P_b K_d + \Theta_w + \Theta_a H^i}$$

Equation 6: Derivation of the Soil Saturation Limit

$$\text{sat} = \frac{S}{P_b} (K_d P_b + \Theta_w + H^i \Theta_a)$$

Equation 7: Derivation of the Particulate Emission Factor (residential and occupation exposures)

$$PEF(m^3/kg) = Q/C \times \frac{3600s/h}{0.036 \times (1-V) (U_m/U_i)^3 \times F(x)}$$

Volatilization factors (VF) are used for volatile chemicals (defined as having a Henry's Law Constant (atm-m³/mol) greater than 10⁻⁵ and a molecular weight less than 200 grams/mol. The VF term in the soil equations is replaced in the equations with a Particulate Emission Factor (PEF) for non-volatile chemicals.

Use of the Volatilization Factor equation to predict vapor-phase concentrations of a chemical in air is not valid if free-product is present. In cases where a chemicals direct-contact screening level exceeds the chemicals theoretical saturation level, and the chemical is a liquid under ambient conditions, the direct-contact screening level is replaced with the chemicals saturation limit.

2.2 Deep Soils

Direct-exposure screening levels for deep soils are calculated based on a construction/trench worker exposure scenario. Exposure assumptions are summarized in Table 1. The assumed exposed skin area and soil ingestion rate are based on guidance presented in the USEPA *Exposure Factor handbook* (USEPA 1997). The inhalation rate, body weight, averaging time and target hazard quotient are set equal to assumptions used in the USEPA Region IX *Preliminary Remediation Goals* (USEPA 2002) for consistency with screening levels for occupational exposure assumptions. The soil adherence factor is taken from trench-worker exposure scenario assumptions developed by the Massachusetts Department of Environmental Protection for use in calculating screening levels for Deep soils (MADEP 1994).

The Massachusetts Department of Environmental Protection assumes exposure durations of three months for noncarcinogens (plus use of subchronic RfDs) and seven years for carcinogens. A seven year (versus three month) exposure duration for carcinogens is used in part because shorter exposure durations were considered to be beyond the limits of cancer risk models. For the purposes of this document, a one-time, three month exposure duration to exposed soils at a site was considered to be inadequate. This may be particularly true for utility workers who re-visit a site numerous times over several years for routine maintenance of underground utilities. As noted in Table 1, a total exposure duration of seven years is assumed for both carcinogens and noncarcinogens. An exposure frequency of 20 days (4 weeks) per year for 7 years yields a total of 140 days total exposure. Construction workers may receive 140 days (roughly 6 months) of exposure in a single year and never visit the site again. Using chronic RfDs (generally less stringent than subchronic RfDs) and spreading the total exposure time over seven years is somewhat conservative but is consistent with the utility worker scenario. A target risk of 1E-06 was used to calculate soil screening levels for carcinogens. A target hazard quotient of 0.2 was used to calculate soil screening levels for noncarcinogens. This is consistent with assumption used to develop screening levels for residential and industrial/commercial exposure scenarios.

"Particulate Emission Factors (PEF)" are intended to relate the concentration of a chemical in soil to the concentration of the chemical in air-born dust. The PEF used for residential and occupational exposure scenarios (1.316E+09 mg-kg/mg/m³) was taken directly from the USEPA *Region IX Preliminary Remediation Goals* guidance document (USEPA 2000). The PEF reflects a concentration of air-born particulate matter of approximately 0.76 ug/m³. This PEF and associated concentration of air-born dust was not considered to be adequately conservative of conditions that may occur at construction sites. A revised PEF for this exposure scenario was derived through use of a "Dust Emission Factor" for construction sites developed by the USEPA. The Dust Emission Factor of 1.2 tons of dust per month, per acre is based on USEPA field studies at apartment complex and commercial center developments in semi-

arid areas (USEPA 1974, 1985). Derivation of the construction-site PEF is summarized in Table 4. The derived PEF (1.44E+06 mg-kg/mg/m³) corresponds to a concentration of air-born dust of approximately 700 ug/m³.

3.0 INDOOR AIR

Target levels for indoor air were calculated based on equations incorporated into the Johnson and Ettinger Model spreadsheet published by the USEPA (USEPA 1997):

Equation 8: Residential Exposures to Carcinogenic Contaminants in Indoor Air

$$C_{air}(ug/m^3) = \frac{TR \times AT_c}{EF_{res} \times ED_{res} \times URF}$$

Equation 9: Occupational Exposures to Carcinogenic Contaminants in Indoor Air

$$C_{air}(ug/m^3) = \frac{TR \times AT_c}{EF_{occ} \times ED_{occ} \times URF}$$

Equation 10: Residential Exposures to Noncarcinogenic Contaminants in Indoor Air

$$C_{air}(ug/m^3) = \frac{THQ \times AT_{nc} \times RfC}{EF_{res} \times ED_{res}}$$

Equation 11: Occupational Exposures to Noncarcinogenic Contaminants in Indoor Air

$$C_{air}(ug/m^3) = \frac{THQ \times AT_{nc} \times RfC}{EF_{occ} \times ED_{occ}}$$

where URF is the unit risk factor carcinogens (ug/m³)⁻¹ for and RfC carcinogens (ug/m³) is the reference concentration for noncarcinogens. A summary of URFs and RfCs for specific chemicals is provided in Table E-3 of Appendix 1.

4.0 DRINKING WATER

Equations for calculation of risk-based drinking water goals are taken from Central Valley Water Board technical document *A Compilation of Water Quality Goals* (RWQCBCV 2000). Default parameter values are noted in Table 1.

Equation 12: Ingestion of Carcinogenic Contaminants in Drinking Water

$$C(ug/L) = \frac{TR \times 70kg}{CSF \times 2L/d} \times 1000ug/mg$$

Equation 13: Ingestion of Noncarcinogenic Contaminants in Drinking Water

$$C(\text{ug/L}) = \frac{RfD \times 70\text{kg}}{2\text{L/d}} \times \frac{RSC}{UF} \times 1000\text{ug/mg}$$

where RSC is the Relative Contribution Factor (default is 20% or 0.2) and UF is an additional uncertainty factor of 10 that is included for Class C carcinogens when no cancer slope factor has been developed.

USEPA Region IX PRGs offer an alternative model for volatile chemicals that takes into account inhalation of vapors during showering and other activities (USEPA 2002):

Equation 14: Ingestion and Inhalation of Carcinogenic Contaminants in Tapwater

$$C(\text{ug/L}) = \frac{TR \times ATc}{EFr[(IFWadj \times CSFo) + (VFw \times InhFadj \times CSFi)]} \times 1000\text{ug/mg}$$

Equation 15: Ingestion and Inhalation of Noncarcinogenic Contaminants in Tapwater

$$C(\text{ug/L}) = \frac{THQ \times BWa \times ATn}{EFr \times EDr \left[\left(\frac{IRWa}{RfDo} \right) + \left(\frac{VFw + IRAa}{RfDi} \right) \right]} \times 1000\text{ug/mg}$$

where VFw is the Volatilization Factor of water to air, assumed to be 0.5 L/m³. A summary of screening levels developed through use of this model is provided in the Table F series of Appendix 1.

REFERENCES

- MADEP, 1994, *Background Documentation for the Development of the MCP Numerical Standards*: Massachusetts Department of Environmental Protection, Bureau of Waste Site Cleanup and Office of Research and Standards, April 1994.
- RWQCBCV, 2000, *A Compilation of Water Quality Goals*: California Environmental Protection Agency, Regional Water Quality Control Board, Central Valley Region, August, 2000, http://www.swrcb.ca.gov/rwqcb5/wq_goals.
- USEPA, 1974, Development of Emission Factors for Fugitive Dust Sources: U.S. Environmental Protection Agency, Publication No. EPA-450/3-74-037, (prepared by Cowherd, C., Axetell, K., Guenther, C., and Jutze, G., Midwest Research Institute).
- USEPA, 1985, Compilation of Air Pollutant Emission Factors, Volume 1: Stationary Point and Area Sources: U.S. Environmental Protection Agency, Office of Air and Radiation, Publication No. AP-42, Fourth Edition, September 1985.
- USEPA, 2000, *User's Guide For The Johnson and Ettinger (1991) Model For Subsurface Vapor Intrusion Into Buildings*: U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, www.epa.gov/oerrpage/superfundrisk/airmodel/.
- USEPA, 2001, Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites: U.S. Environmental Protection Agency, Solid Waste and Emergency Response, OSWER 9355.4-24, March 2001.
- USEPA, 2002, Preliminary Remediation Goals: U.S. Environmental Protection Agency, Region IX, October 2002, www.epa.gov/region09/waste/sfund/prg/intro.htm.

**TABLE 1. HUMAN EXPOSURE PARAMETER DEFINITIONS
AND DEFAULT VALUES**

Symbol	Definition (units)	Default	References (refer to USEPA 2000 for full references)
CSFo	Cancer slope factor oral (mg/kg-d) ⁻¹	--	Chemical specific - Appendix 1, Table J
CSFi	Cancer slope factor inhaled (mg/kg-d) ⁻¹	--	Chemical specific - Appendix 1, Table J
RfDo	Reference dose oral (mg/kg-d)	--	Chemical specific - Appendix 1, Table J
RfDi	Reference dose inhaled (mg/kg-d)	--	Chemical specific - Appendix 1, Table J
TRr/o	Target cancer risk - residential, occupational/ industrial exposure scenario	10 ⁻⁶	USEPA 2000
*TRctw	Target cancer risk - construction/trench worker exposure scenario	10 ⁻⁶	model assumption
THQ	Target hazard quotient	0.2	modified from USEPA 2000
BWa	Body weight, adult (kg)	70	RAGS (Part A), USEPA 1989 (EPA/540/1-89/002) Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
BWc	Body weight, child (kg)	15	
ATc	Average time - carcinogens (days)	25,550	RAGS (Page A), USEPA 1989 (EPA/540/1-89/002)
ATn	Average time - noncarcinogens (days)	ED*365	USEPA 2000
SAar	Exposed surface area, adult res. (cm ² /day)	5,700	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005)
SAaw	Exposed surface area, adult occ. (cm ² /day)	3,300	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005))
SAC	Exposed surface area, child (cm ² /day)	2,800	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005))
*SAac/tw	Exposed surface area, construction/trench worker (cm ² /day)	5,800	Exposure Factors, USEPA 1997 (EPA/600/P-95/002Fa)
AFar	Adherence factor, adult res. (mg/cm ²)	0.07	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005)
AFaw	Adherence factor, occupational (mg/cm ²)	0.20	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005)
*AFctw	Adherence factor, construction/trench worker (mg/cm ²)	0.51	Massachusetts DEP (1994)
AFc	Adherence factor, child (mg/cm ²)	0.20	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005)
ABS	Skin absorption (unitless): chemical specific	--	Dermal Assessment, USEPA 2000 (EPA/540/R-99/005)
IRaA	Inhalation rate - adult (m ³ /day)	20	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
IRAc	Inhalation rate - child (m ³ /day)	10	Exposure Factors, USEPA 1997 (EPA/600/P-95/002Fa)
*IRActw	Inhalation rate - construction/trench worker (m ³ /day)	20	Exposure Factors, USEPA 1997 (EPA/600/P-95/002Fa)
IRWa	Drinking water ingestion - adult (L/day)	2	RAGS (Part A), USEPA 1989 (EPA/540/1-89/002)
IRWc	Drinking water ingestion - child (L/day)	1	PEA Cal-EPA (DTSC, 1994)
IRSa	Soil ingestion - adult (mg/day)	100	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
IRSc	Soil ingestion - child (mg/day)	200	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
IRSo	Soil ingestion - occupational (mg/day)	50	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
*IRSctw	Soil ingestion - construction/trench worker (mg/day)	330	USEPA 2001
EFr	Exposure frequency - residential (d/y)	350	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
EFo	Exposure frequency - occupational (d/y)	250	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
*EFctw	Exposure frequency - construction/trench worker (d/y)	20	Massachusetts DEP (1994)
EDr	Exposure duration - residential (years)	30	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
EDc	Exposure duration - child (years)	6 ^a	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
EDo	Exposure duration - occupational (years)	25	Exposure Factors, USEPA 1991 (OSWER No. 9285.6-03)
*EDctw	Exposure duration - construction/trench worker (years)	7	modified from Massachusetts DEP (1994)
IFSadj	Ingestion factor, soils ([mg-yr]/[kg-d])	114	RAGS (Part B, v 1991 (OSWER No. 9285.7-01B)
SFSadj	Skin contact factor, soils ([mg-yr]/[kg-d])	361	By analogy to RAGS (Part B)
InhFadj	Inhalation factor ([m ³ -yr]/[kg-d])	11	By analogy to RAGS (Part B)
IFWadj	Ingestion factor, water ([l-yr]/[kg-d])	1.1	By analogy to RAGS (Part B)
VFw	Volatilization factor for water (L/m ³)	0.5	RAGS (Part B), USEPA 1991 (OSWER No. 9285.7-01B)
PEFres/oc	Particulate emission factor (m ³ /kg) - residential/occupational exposure scenarios	1.32E+09	Soil Screening Guidance (USEPA 1996a)
*PEFctw	Particulate emission factor (m ³ /kg) - construction/trench worker exposure scenarios	1.44E+06	Based on Construction Site Dust Emission Factors (USEPA 1974, 1985). See attached table.
VFs	Volatilization factor for soil (m ³ /kg)	-	Chemical specific; Soil Screening Guidance (USEPA 1996a,b)
sat	Soil saturation concentration (mg/kg)	-	Chemical specific; Soil Screening Guidance (USEPA 1996a,b)

Primary Reference: USEPA, 2002, *Preliminary Remediation Goals*: U.S. Environmental Protection Agency, Region IX, October 2002,

a Exposure duration for lifetime residents is assumed to be 30 years total. For carcinogens, exposures are combined for children (6 years) and adults (24 years).

* This document only. Not presented in USEPA Region IX PRGs.

**TABLE 2. VOLATILIZATION FACTOR PARAMETER DEFINITIONS
AND DEFAULT VALUES**

Parameter	Definition (units)	Default
VF_s	Volatilization factor M^3/kg	--
D_A	Apparent diffusivity (cm^2/s)	--
Q/C	Inverse of the mean conc. at the center of a 0.5-acre square source ($g/m^2 \cdot s$ per kg/m^3)	68.81
T	Exposure interval (s)	9.5×10^8
ρ_{ob}	Dry soil bulk density (g/cm^3)	1.5
θ_{a_a}	Air filled soil porosity (L_{air}/L_{soil})	0.28 or n-w
n	Total soil porosity (L_{pore}/L_{soil})	0.43 or $1 - (b/s)$
θ_{a_w}	Water-filled soil porosity (L_{water}/L_{soil})	0.15
ρ_{os}	Soil particle density (g/cm^3)	2.65
D_i	Diffusivity in air (cm^2/s)	Chemical-specific
H	Henry's Law constant ($atm \cdot m^3/mol$)	Chemical-specific
H'	Dimensionless Henry's Law constant	Calculated from H by multiplying by 41 (USEPA 1991a)
D_w	Diffusivity in water (cm^2/s)	Chemical-specific
K_d	Soil-water partition coefficient (cm^3/g) = $K_{oc} \times f_{oc}$	Chemical-specific
K_{oc}	Soil organic carbon-water partition coefficient (cm^3/g)	Chemical-specific
f_{oc}	Fraction organic carbon in soil (g/g)	0.006 (0.6%)

**TABLE 3. PARTICULATE EMISSION FACTOR PARAMETER DEFINITIONS AND
DEFAULT VALUES - RESIDENTIAL/OCCUPATIONAL SCENARIOS**

Parameter	Definition (units)	Default
PEF	Particulate emission factor (m ³ /kg)	1.316 x 10 ⁹
Q/C	Inverse of the mean concentration at the center of a 0.5-acre-square source (g/m ² -s per kg/m ³)	90.80
V	Fraction of vegetative cover (unitless)	0.5
Um	Mean annual windspeed (m/s)	4.69
Ut	Equivalent threshold value of windspeed at 7 m (m/s) 11.32	11.32
F(x)	Function dependent on Um/Ut derived using Cowherd (1985) (unitless)	0.194

**TABLE 4. PARTICULATE EMISSION FACTOR FOR
CONSTRUCTION/TRENCH WORKER EXPOSURE SCENARIO**

Dust Generated (moderate to heavy construction) (M_{dust}):			
Dust Emission Factor (EF):	1.2 2400 1089	tons/mo-acre lbs/mo-acre kgs/mo-acre	USEPA 1974, 1985 conversion conversion
Volume Air Passing Over Site Per Month Per Acre (V_{air}):			
Length Perpendicular To Wind (L):	1 43560 4047 64	acre ft ² m ² m	Default EF area conversion conversion $L = \text{Area}^{0.5}$
Air Mixing Zone Height (MZ):	2	m	model assumption
Ave Wind Speed (V):	4.69	m/s	USEPA 2000 (default PRG value)
Seconds per 30.4 Day Month (S):	2.63E+06	sec/month	conversion
Volume Air (Volume-air):	1.57E+09	m ³	Volume-air = $L \times MZ \times V \times S$
Average Concentration Dust in Air ($C_{\text{dust-air}}$):			
Concentration Dust ($C_{\text{dust-air}}$):	6.95E-07 0.695	kg/m ³ mg/m ³	($C_{\text{air}} = M_{\text{dust}} / \text{Volume-air}$) conversion
Particulate Emission Factor (PEF):			
Concentration soil in dust ($C_{\text{dust-soil}}$):	1,000,000	mg/kg	Model assumption - 100% (1000000 mg/kg) of dust is derived from on-site soil.
PEF:	1.44E+06	(mg/kg)/ (mg/m ³)	$\text{PEF} = C_{\text{dust-soil}} / C_{\text{dust-air}}$

Attachment

Text of USEPA Region IX Preliminary Remediation Goals Document (October 2002)



UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION IX
75 Hawthorne Street
San Francisco, CA 94105

October 1, 2002

Subject: Region 9 PRGs Table 2002 Update

From: Stanford J. Smucker, Ph.D.
Regional Toxicologist (SFD-8-B)
Technical Support Team

To: PRGs Table Users

With this cover letter, we announce the update to the Region 9 PRGs table for 2002. The PRGs table contains over 600 preliminary remediation goals (PRGs) for contaminants in soil, air, and tap water. Region 9 PRGs are risk-based concentrations that are intended to assist risk assessors and others in initial screening-level evaluations of environmental measurements.

As their name implies, Region 9 PRGs may also be viewed as preliminary cleanup goals for an individual chemical, but in this context, they are best viewed as dynamic and subject to change because they are generic and based on direct contact exposures which may not address site-specific conditions and/or indirect exposure pathways at sites (See Exhibit 1-1 in "Region 9 PRGs Table Users Guide/Technical Background Document"). Also for planning purposes, these human health based PRGs should always be considered in conjunction with ARAR-based PRGs (e.g. MCLs), ecological benchmarks, and "background" conditions before establishing a final cleanup level for a particular site.

You can find the PRGs 2002 table, InterCalc tables, "Region 9 PRGs Table Users Guide/Technical Background Document", and additional helpful toxicological and risk assessment information at:

<http://www.epa.gov/region09/waste/sfund/prg/> .

We view risk-based PRGs as "evergreen". Ongoing changes to the PRGs reflect continuing improvements in our scientific knowledge base and state-of-the-art approaches to risk assessment. In the new *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (Supplemental SSL Guidance, EPA 2001a), two different soil ingestion rates are assumed for non-construction workers: 100 mg/day is assumed for outdoor workers whereas 50 mg/day is assumed for indoor workers. The default value of 100 mg/day for outdoor workers is also recommended by EPA's Technical Review Workgroup for Lead (TRW), and it reflects increased exposures to soils for outdoor workers relative to their indoor counterparts. For more on this, please see Section 4.1 of the "Region 9 PRGs Table Users Guide/Technical Background Document" or refer to the Supplemental SSL Guidance available at the following website:

<http://www.epa.gov/superfund/resources/soil/index.htm>

Because the Region 9 PRGs are generic and intended for screening sites early in the investigation process (often before site-specific information is available), we have chosen to use the 100 mg/day soil ingestion (i.e. outdoor worker) assumption to calculate industrial soil PRGs. Please note that previous issues of the Region 9 PRGs table assumed 50 mg/day soil ingestion rate for workers. This change in soil ingestion rates is reflected in a somewhat lower (more stringent) industrial soils PRG for many contaminants. The appropriateness of this assumption for a particular site may be evaluated when additional information becomes available regarding site conditions or site development.

In addition to changes in exposure factor assumptions, several chemicals have new or revised toxicity values that results in changes to the PRG calculations. To facilitate the users review, chemicals with new and revised toxicological criteria are presented in bold in the 2002 table and also listed here for convenience: **acetonitrile, benzyl chloride, boron, bromate, 1,3-butadiene, 1-butanol, butylbenzenes, cacodylic acid, cadmium (California State value), chloroform, chloronitrobenzenes, chrysene (California State value), cobalt, 1,2-dibromo-3-chloropropane (California State value), 1,1-dichloroethylene, diethylene glycol ethers, diethylformamide, dinitrobenzenes, di-n-octyl phthalate, diphenyl sulfone, ethylbenzene, HCH, hexachlorocyclopentadiene, kepone, lead (California State value), MTBE, 2-nitroaniline, carcinogenic PAHs, perchlorate, polychlorinated terphenyls, benzo(k)fluoranthene (California State value), propylbenzene, propylene glycol, quinoline, tetrachloroethylene, tetrahydrofuran, thiocyanate, 1,1,1-trichloroethane, trichloroethylene, 2,4,6-trichlorophenol, 1,2,3-trichloropropane, triphenylphosphine oxide, tris(2-chloroethyl) phosphate, vinyl chloride, and xylene.**

Also in this update to the "Region 9 PRGs Table User's Guide/Technical Background Document", we have added a brief discussion of special case chemicals for which an alternate approach was applied in the derivation of the Region 9 PRGs (Section 2.3). Increasingly, chemical-specific approaches are being used that do not lend themselves to a single PRG model. Special case chemicals that are discussed include: cadmium, chromium 6, lead, manganese, nitrate/nitrite, thallium, and vinyl chloride.

Finally it should be recognized by all that use the PRGs table that not all PRG values in the table are "created equal". For some chemicals, a robust data set exists upon which the toxicological criteria are based whereas for others, there may be relatively few studies that form the basis of the PRG calculation. Also, PRGs for some chemicals are based on withdrawn toxicity values or route-extrapolated values. Withdrawn and route-extrapolated numbers are shown in the table because we still need to deal with these contaminants during the long delays before replacement numbers are ready. Please consult with your toxicologist or agency risk assessor to best address potential uncertainties associated with chemical-specific PRGs, especially if the chemical is a risk driver at your site.

As with any risk-based tool, there exists the potential for misuse. We try to highlight potential problems in Section 3.8. However, it should be noted that the use of PRGs at a particular site becomes the responsibility of the user. It is recommended that the user verify the numbers with an agency toxicologist or risk assessor because the toxicity / exposure information in the table may contain errors or default assumptions that need to be refined based on further evaluation. If you find an error please send me a note via email at smucker.stan@epa.gov.

DISCLAIMER

Preliminary remediation goals (PRGs) focus on common exposure pathways and may not consider all exposure pathways encountered at CERCLA / RCRA sites (Exhibit 1-1). PRGs do not consider impact to groundwater or address ecological concerns. The PRG table is specifically not intended as a (1) stand-alone decision-making tool, (2) as a substitute for EPA guidance for preparing baseline risk assessments, (3) a rule to determine if a waste is hazardous under RCRA, or (4) set of final cleanup or action levels to be applied at contaminated sites.

The guidance set out in this document is not final Agency action. It is not intended, nor can it be relied upon to create any rights enforceable by any party in litigation with the United States. EPA officials may decide to follow the guidance provided herein, or act at variance with the guidance, based on an analysis of specific circumstances. The Agency also reserves the right to change this guidance at any time without public notice.

1.0 INTRODUCTION

Region 9 Preliminary Remediation Goals (PRGs) are risk-based tools for evaluating and cleaning up contaminated sites. They are being used to streamline and standardize all stages of the risk decision-making process.

The Region 9 PRG table combines current EPA toxicity values with "standard" exposure factors to estimate contaminant concentrations in environmental media (soil, air, and water) that the agency considers protective of humans (including sensitive groups), over a lifetime. Chemical concentrations above these levels would not automatically designate a site as "dirty" or trigger a response action. However, exceeding a PRG suggests that further evaluation of the potential risks that may be posed by site contaminants is appropriate. Further evaluation may include additional sampling, consideration of ambient levels in the environment, or a reassessment of the assumptions contained in these screening-level estimates (e.g. appropriateness of route-to-route extrapolations, appropriateness of using chronic toxicity values to evaluate childhood exposures, appropriateness of generic exposure factors for a specific site etc.).

The PRG concentrations presented in the table can be used to screen pollutants in environmental media, trigger further investigation, and provide an initial cleanup goal if applicable. When considering PRGs as cleanup goals, residential concentrations should be used for maximum beneficial uses of a property. Industrial concentrations are included in the table as an alternative cleanup goal for soils. **In general, it is recommended that industrial PRGs not be used for screening sites unless they are used in conjunction with residential values.**

Before applying PRGs as screening tools or initial goals, the user of the table should consider whether the exposure pathways and exposure scenarios at the site are fully accounted for in the PRG calculations. Region 9 PRG concentrations are based on direct contact pathways for which generally accepted methods, models, and assumptions have been developed (i.e. ingestion, dermal contact, and inhalation) for specific land-use conditions and do not consider impact to groundwater or ecological receptors (see Developing a Conceptual Site Model below).

EXHIBIT 1-1
TYPICAL EXPOSURE PATHWAYS BY MEDIUM
FOR RESIDENTIAL AND INDUSTRIAL LAND USES^a

EXPOSURE PATHWAYS, ASSUMING:		
MEDIUM	RESIDENTIAL LAND USE	INDUSTRIAL LAND USE
Ground Water	<i>Ingestion from drinking</i>	Ingestion from drinking
	<i>Inhalation of volatiles</i>	Inhalation of volatiles
	Dermal absorption from bathing	Dermal absorption
Surface Water	<i>Ingestion from drinking</i>	Ingestion from drinking
	<i>Inhalation of volatiles</i>	Inhalation of volatiles
	Dermal absorption from bathing	Dermal absorption
	Ingestion during swimming	
	Ingestion of contaminated fish	
Soil	<i>Ingestion</i>	<i>Ingestion</i>
	<i>Inhalation of particulates</i>	<i>Inhalation of particulates</i>
	<i>Inhalation of volatiles</i>	<i>Inhalation of volatiles</i>
	Exposure to indoor air from soil gas	Exposure to indoor air from soil gas
	Exposure to ground water contaminated by soil leachate	Exposure to ground water contaminated by soil leachate
	Ingestion via plant, meat, or dairy products	Inhalation of particulates from trucks and heavy equipment
	<i>Dermal absorption</i>	<i>Dermal absorption</i>

Footnote:

^aExposure pathways considered in the PRG calculations are indicated in boldface italics.

2.0 READING THE PRG TABLE

2.1 General Considerations

With the exceptions described below, PRGs are chemical concentrations that correspond to fixed levels of risk (i.e. either a one-in-one million [10^{-6}] cancer risk or a noncarcinogenic hazard quotient of 1) in soil, air, and water. In most cases, where a substance causes both cancer and noncancer (systemic) effects, the 10^{-6} cancer risk will result in a more stringent criteria and consequently this value is presented in the printed copy of the table. PRG concentrations that equate to a 10^{-6} cancer risk are indicated by "ca". PRG concentrations that equate to a hazard quotient of 1 for noncarcinogenic concerns are indicated by "nc".

If the risk-based concentrations are to be used for site screening, it is recommended that both cancer and noncancer-based PRGs be used. Both carcinogenic and noncarcinogenic values may be obtained at the Region 9 PRG homepage at:

<http://www.epa.gov/region09/waste/sfund/prg/>

It has come to my attention that some users have been multiplying the cancer PRG concentrations by 10 or 100 to set "action levels" for triggering remediation or to set less stringent cleanup levels for a specific site after considering non-risk-based factors such as ambient levels, detection limits, or technological feasibility. This risk management practice recognizes that there may be a range of values that may be "acceptable" for carcinogenic risk (EPA's risk management range is one-in-a-million [10^{-6}] to one-in-ten thousand [10^{-4}]). However, this practice could lead one to overlook serious noncancer health threats and it is strongly recommended that the user consult with a toxicologist or regional risk assessor before doing this. For carcinogens, I have indicated by asterisk ("ca*") in the PRG table where the noncancer PRGs would be exceeded if the cancer value that is displayed is multiplied by 100. Two stars ("ca**") indicate that the noncancer values would be exceeded if the cancer PRG were multiplied by 10. There is no range of "acceptable" noncarcinogenic "risk" so that under no circumstances should noncancer PRGs be multiplied by 10 or 100, when setting final cleanup criteria. In the rare case where noncancer PRGs are more stringent than cancer PRGs set at one-in-one-million risk, a similar approach has been applied (e.g. "nc**").

In general, PRG concentrations in the printed table are risk-based but for soil there are two important exceptions: (1) for several volatile chemicals, PRGs are based on the soil saturation equation ("sat") and (2) for relatively less toxic inorganic and semivolatile contaminants, a non-risk based "ceiling limit" concentration is given as 10^{+5} mg/kg ("max"). At the Region 9 PRG website, the risk-based calculations for these same chemicals are also available in the "InterCalc Tables" if the user wants to view the risk-based concentrations prior to the application of "sat" or "max". For more information on why the "sat" value and not a risk-based value is presented for several volatile chemicals in the PRGs table, please see the discussion in Section 4.5.

With respect to applying a "ceiling limit" for chemicals other than volatiles, it is recognized that this is not a universally accepted approach. Some within the agency argue that all values should be risk-based to allow for scaling (for example, if the risk-based PRG is set at a hazard quotient = 1.0, and the user would like to set the hazard quotient to 0.1 to take into account multiple chemicals, then this is as simple as multiplying the risk-based PRG by 1/10th). If scaling is necessary, PRG users can do

this simply by referring to the “InterCalc Tables” at our website where risk-based soil concentrations are presented for all chemicals (see soil calculations, “combined” pathways column).

In spite of the fact that applying a ceiling limit is not a universally accepted approach, we have opted to continue applying a “max” soil concentration to the PRGs table for the following reasons:

- Risk-based PRGs for some chemicals in soil exceed unity (>1,000,000 mg/kg) which is not possible.
- The ceiling limit of 10^{+5} mg/kg is equivalent to a chemical representing 10% by weight of the soil sample. At this contaminant concentration (and higher), the assumptions for soil contact may be violated (for example, soil adherence and windborne dispersion assumptions) due to the presence of the foreign substance itself.
- PRGs currently do not address short-term exposures (e.g. pica children and construction workers). Although extremely high soil PRGs are likely to represent relatively non-toxic chemicals, such high values may not be justified if in fact more toxicological data were available for evaluating short-term and/or acute exposures.

In addition to Region 9 PRG values, the PRGs table also includes California EPA PRGs (“CAL-Modified PRGs”) for specific chemicals where CAL-EPA screening values may be “significantly” more restrictive than the federal values (see Section 2.4) and EPA OSWER soil screening levels (SSLs) for protection of groundwater (see Section 2.5).

2.2 Toxicity Values

Hierarchy of Toxicity Values

EPA toxicity values, known as noncarcinogenic reference doses (RfD) and carcinogenic slope factors (SF) were obtained from IRIS, NCEA through September 2002, and HEAST (1997). The priority among sources of toxicological constants in order of preference is as follows: (1) IRIS (indicated by “i”), (2) NCEA (“n”), (3) HEAST (“h”), (4) withdrawn from IRIS or HEAST and under review (“x”) or obtained from other EPA documents (“o”). This hierarchy is subject to change once the HEAST tables are updated.

Inhalation Conversion Factors

As of January 1991, IRIS and NCEA databases no longer present RfDs or SFs for the inhalation route. These criteria have been replaced with reference concentrations (RfC) for noncarcinogenic effects and unit risk factors (URF) for carcinogenic effects. However, for purposes of estimating risk and calculating risk-based concentrations, inhalation reference doses (RfDi) and inhalation slope factors (SF_i) are preferred. This is not a problem for most chemicals because the inhalation toxicity criteria are easily converted. To calculate an RfDi from an RfC, the following equation and assumptions may be used for most chemicals:

$$\text{RfDi} \frac{\text{mg}}{(\text{kg} \cdot \text{day})} = \text{RfC} (\text{mg} / \text{m}^3) \times \frac{20 \text{m}^3}{\text{day}} \times \frac{1}{70 \text{kg}}$$

Likewise, to calculate an SF_i from an inhalation URF, the following equation and assumptions may be used:

$$SFi \frac{(\text{kg} \cdot \text{day})}{(\text{mg})} = \text{URF} (\text{m}^3 / \text{ug}) \times \frac{\text{day}}{20\text{m}^3} \times 70\text{kg} \times \frac{10^3 \text{ ug}}{\text{mg}}$$

Substances with New or Withdrawn Toxicity Values

To help users rapidly identify substances with new or revised toxicity values, these chemicals are listed in boldface type in the PRGs table. This issue of the table contains new or revised toxicity values for: **acetonitrile, benzyl chloride, boron, bromate, 1,3-butadiene, 1-butanol, butylbenzenes, cacodylic acid, cadmium (California State value), chloroform, chloronitrobenzenes, chrysene (California State value), cobalt, 1,2-dibromo-3-chloropropane (California State value), 1,1-dichloroethylene, diethylene glycol ethers, diethylformamide, dinitrobenzenes, di-n-octyl phthalate, diphenyl sulfone, ethylbenzene, HCH, hexachlorocyclopentadiene, kepone, lead (California State value), MTBE, 2-nitroaniline, carcinogenic PAHs, perchlorate, polychlorinated terphenyls, benzo(k)fluoranthene (California State value), propylbenzene, propylene glycol, quinoline, tetrachloroethylene, tetrahydrofuran, thiocyanate, 1,1,1-trichloroethane, trichloroethylene, 2,4,6-trichlorophenol, 1,2,3-trichloropropane, triphenylphosphine oxide, tris(2-chloroethyl) phosphate, vinyl chloride, and xylene.**

Chemicals that have been delisted because they are outdated, undocumented, or derived from a data base other than IRIS, HEAST or NCEA include: acifluorfen, 4-bromophenyl phenyl ether, chloroacetaldehyde, 2-chloroethyl vinyl ether, hexachlorodibenzo-p-dioxin mixture (HxCDD), maneb, methyl chlorocarbonate, nitrapyrin, nitric oxide, and 4-nitrophenol.

Route-to-Route Methods

Route-to-route extrapolations ("r") were frequently used when there were no toxicity values available for a given route of exposure. Oral cancer slope factors ("SFO") and reference doses ("RfDo") were used for both oral and inhaled exposures for organic compounds lacking inhalation values. Inhalation slope factors ("SFI") and inhalation reference doses ("RfDi") were used for both inhaled and oral exposures for organic compounds lacking oral values. Route extrapolations were not performed for inorganics due to portal of entry effects and known differences in absorption efficiency for the two routes of exposure.

An additional route extrapolation is the use of oral toxicity values for evaluating dermal exposures. For many chemicals, a scientifically defensible data base does not exist for making an adjustment to the oral slope factor/RfD to estimate a dermal toxicity value. Based on the current guidance (USEPA 2001b), the only chemical for which an adjustment is recommended is cadmium. An oral absorption efficiency of 5% is assumed for cadmium which leads to an estimated dermal reference dose (RfDd) of 2.5E-05 that was used in the soil PRG calculations for cadmium.

Although route-to-route methods may be a useful screening procedure, the appropriateness of these default assumptions for specific contaminants should be verified by a toxicologist or regional risk assessor. Please note that whenever route-extrapolated values are used to calculate risk-based PRGs, additional uncertainties are introduced in the calculation.

2.3 Region 9 PRGs Derived with Special Considerations

Most of the Region 9 PRGs are readily derived by referring to Equations 4-1 thru 4-8 contained in this "User's Guide/Technical Background Document" to the Region 9 PRGs. However, there are some chemicals for which the standard equations do not apply and/or adjustments to the toxicity values are recommended. These special case chemicals are discussed below.

Cadmium The PRGs for Cadmium are based on the oral RfD for water which is slightly more conservative (by a factor of 2) than the RfD for food. Because the PRGs are considered screening values, we elected to use the more conservative RfD for cadmium. However, reasonable arguments could be made for applying an RfD for food (instead of the oral RfD for water) for some media such as soils.

The water RfD for cadmium assumes a 5% oral absorption factor. The assumption of an oral absorption efficiency of 5% for Cadmium leads to an estimated dermal RfD of $2.5E-05$. The PRG calculations incorporate these adjustments per recent guidance (USEPA 2001b).

Chromium 6 For Chromium 6 (Cr6), IRIS shows an air unit risk of $1.2E-2$ per (ug/cu.m) or expressed as an inhalation cancer slope factor (adjusting for inhalation/body weight) of $42 \text{ (mg/kg-day)}^{-1}$. However, the supporting documentation in the IRIS file states that these toxicity values are based on an assumed 1:6 ratio of Cr6:Cr3. Because of this assumption, we in Region 9 prefer to present PRGs based on these cancer toxicity values as "total chromium" numbers.

In the PRG tables, we also include a Cr6 specific value (assuming 100% Cr6) that is derived by multiplying the "total chromium" value by 7, yielding a cancer potency factor of $290 \text{ (mg/kg-day)}^{-1}$. This is considered to be an overly conservative assumption by some within the Agency. However, this calculation is also consistent with the State of California's interpretation of the Mancuso study that forms the basis of Cr6's toxicity values.

If you are working on a project outside of California (and outside of Region 9), you may want to contact the appropriate regulatory officials to determine what their position is on this issue. As mentioned, Region 9 also includes PRGs for "total chromium" which is based on the same ratio (1:6 ratio Cr6:Cr3) that forms the basis of the cancer slope factor of $42 \text{ (mg/kg-day)}^{-1}$ presented in IRIS.

Lead Residential PRGs for Lead (Region 9 EPA and California EPA) are derived based on pharmacokinetic models. Both EPA's Integrated Exposure Uptake Biokinetic (IEUBK) Model and California's LeadSpread model are designed to predict the probable blood lead concentrations for children between six months and seven years of age who have been exposed to lead through various sources (air, water, soil, dust, diet and *in utero* contributions from the mother). Run in the reverse, these models also allow the user to calculate lead PRGs that are considered "acceptable" by EPA or the State of California.

The California LeadSpread model can also estimate PRGs for non-residential exposures (e.g. worker) whereas EPA uses a second Adult Lead Model to estimate PRGs for an industrial setting.

For more information on EPA's Lead models used to estimate residential and industrial PRGs, please refer to the following website:

<http://www.epa.gov/ocrrpage/superfund/programs/lead/>

For more information on California's LeadSpread Model and Cal-Modified PRGs for lead, please go to:

<http://www.dtsc.ca.gov/ScienceTechnology/ledspread.html>

Manganese The IRIS RfD (0.14 mg/kg-day) includes manganese from all sources, including diet. The author of the IRIS assessment for manganese recommends that the dietary contribution from the normal U.S. diet (an upper limit of 5 mg/day) be subtracted when evaluating non-food (e.g. drinking water or soil) exposures to manganese, leading to a RfD of 0.071 mg/kg-day for non-food items. The explanatory text in IRIS further recommends using a modifying factor of 3 when calculating risks associated with non-food sources due to a number of uncertainties that are discussed in the IRIS file for manganese, leading to a RfD of 0.024 mg/kg-day. This modified RfD is applied in the derivation of the Region 9 PRGs for soil and water. For more information regarding the Manganese RfD, you may want to contact Dr. Bob Benson at (303) 312-7070.

Nitrates/Nitrites Tap water PRGs for Nitrates/Nitrites are based on the MCL as there is no available RfD for these compounds. For more information, please see IRIS at:
<http://www.epa.gov/iriswebp/iris/index.html>

Thallium IRIS has many values for the different salts of thallium. However, our analytical data packages typically report "thallium". Therefore, as a practical matter it makes more sense to report a PRG for plain thallium. We have done this by making the adjustment contained in the IRIS file for thallium sulfate based on the molecular weight of the thallium in the thallium salt. The adjusted oral RfD for plain thallium is 6.6 E-05 mg/kg-day which we use to calculate a thallium PRG.

Vinyl Chloride In EPA's recent reassessment of vinyl chloride toxicity, IRIS presents two cancer slope factors for vinyl chloride (VC): one that is intended to be applied towards evaluating adult risks and a second more protective slope factor that takes into account the unique susceptibility of developing infants and young children. For residential PRGs, the Region 9 PRGs table applies the more conservative cancer potency factor that addresses exposures to both children and adults whereas for the industrial soils PRG, the adult only cancer slope factor is applied.

Because of the age-dependent vulnerability associated with vinyl chloride exposures, and due to the method that is applied in deriving the cancer slope factor for VC, an assumption of a 70 year exposure over the lifetime is assumed, consistent with the way that the toxicity value for VC was derived. Therefore, instead of the usual exposure assumption of 6 years as a child and 24 years as an adult that is assumed for carcinogenic substances, we have revised the exposure assumption for VC to 6 years as a child and 64 years as adult. Since most of the cancer risk is associated with the first 30 years of exposure to VC, there is actually little difference between a 30 year exposure assumption (typically assumed for Superfund risk assessments) and the 70 year exposure assumption that is assumed in calculating the PRG for VC.

2.4 "Cal-Modified PRGs"

When EPA Region 9 first came out with a Draft of the PRGs table in 1992, there was concern expressed by California EPA's Department of Toxic Substances and Control (DTSC) that for some chemicals the risk-based concentrations calculated using Cal-EPA toxicity values were "significantly" more protective than the risk-based PRGs calculated by Region 9. At an interagency meeting

comprised of mostly toxicologists, it was agreed that PRG values are at best order-of-magnitude estimates, so that if we assume a logarithmic scale, then a difference greater than 3.3 ($\frac{1}{2}$ log above or below) would be considered a significant difference. Therefore, for individual chemicals where California PRG values are significantly more protective than Region 9 EPA PRGs, Cal-Modified PRGs are included in the Region 9 PRGs table. For more information on Cal-Modified PRGs, the reader may want to contact Dr. Michael Wade in Cal-EPA's Department of Toxic Substances (DTSC) at (916) 255-6653.

Please note that in the State of California, Cal-Modified PRGs should be used as screening levels for contaminated sites because they are more stringent than the Federal numbers.

2.5 Soil Screening Levels

Generic, soil screening levels (SSLs) for the protection of groundwater have been included in the PRG table for 100 of the most common contaminants at Superfund sites. Generic SSLs are derived using default values in standardized equations presented in EPA OSWER's *Soil Screening Guidance* series, available on the web at <http://www.epa.gov/superfund/resources/soil/index.htm>.

The SSLs were developed using a default dilution-attenuation factor (DAF) of 20 to account for natural processes that reduce contaminant concentrations in the subsurface. Also included are generic SSLs that assume no dilution or attenuation between the source and the receptor well (i.e., a DAF of 1). These values can be used at sites where little or no dilution or attenuation of soil leachate concentrations is expected at a site (e.g., sites with shallow water tables, fractured media, karst topography, or source size greater than 30 acres).

In general, if an SSL is not exceeded for the migration to groundwater pathway, the user may eliminate this pathway from further investigation.

It should be noted that in the State of California, the California Regional Water Quality Control Board has derived "California SSLs" for a number of pathways including migration to groundwater. These are not included in the Region 9 PRGs table, but may be accessed at the following website:

<http://www.swrcb.ca.gov/rwqcb2/rbsl.htm>

Or, for more information on the "California SSLs", please contact Dr Roger Brewer at: (510) 622-2374.

2.6 Miscellaneous

Volatile organic compounds (VOCs) are indicated by "1" in the VOC column of the table and in general, are defined as those chemicals having a Henry's Law constant greater than 10^{-5} (atm-m³/mol) and a molecular weight less than 200 g/mole). Three borderline chemicals (dibromochloromethane, 1,2-dibromochloropropane, and pyrene) which do not strictly meet these criteria of volatility have also been included based upon discussions with other state and federal agencies and after a consideration of vapor pressure characteristics etc. Volatile organic chemicals are evaluated for potential volatilization from soil/water to air using volatilization factors (see Section 4.1).

Chemical-specific dermal absorption values for contaminants in soil and dust are presented for arsenic, cadmium, chlordane, 2,4-D, DDT, lindane, TCDD, PAHs, PCBs, and pentachlorophenols as recommended in the "Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment) Interim Guidance" (USEPA 2001b). Otherwise, default skin absorption fractions are assumed to be 0.10 for nonvolatile organics. Please note that previous defaults of 0.01 and 0.10 for inorganics and VOCs respectively, have been withdrawn per new guidance.

3.0 USING THE PRG TABLE

The decision to use PRGs at a site will be driven by the potential benefits of having generic risk-based concentrations in the absence of site-specific risk assessments. The original intended use of PRGs was to provide initial cleanup goals for individual chemicals given specific medium and land-use combinations (see RAGS Part B, 1991), however risk-based concentrations have several applications. They can also be used for:

- Setting health-based detection limits for chemicals of potential concern
- Screening sites to determine whether further evaluation is appropriate
- Calculating cumulative risks associated with multiple contaminants

A few basic procedures are recommended for using PRGs properly. These are briefly described below. Potential problems with the use of PRGs are also identified.

3.1 Developing a Conceptual Site Model

The primary condition for use of PRGs is that exposure pathways of concern and conditions at the site match those taken into account by the PRG framework. Thus, it is always necessary to develop a conceptual site model (CSM) to identify likely contaminant source areas, exposure pathways, and potential receptors. This information can be used to determine the applicability of PRGs at the site and the need for additional information. For those pathways not covered by PRGs, a risk assessment specific to these additional pathways may be necessary. Nonetheless, the PRG lookup values will still be useful in such situations for focusing further investigative efforts on the exposure pathways not addressed.

To develop a site-specific CSM, perform an extensive records search and compile existing data (e.g. available site sampling data, historical records, aerial photographs, and hydrogeologic information). Once this information is obtained, CSM worksheets such as those provided in ASTM's *Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites* (1995) can be used to tailor the generic worksheet model to a site-specific CSM. The final CSM diagram represents linkages among contaminant sources, release mechanisms, exposure pathways and routes and receptors. It summarizes our understanding of the contamination problem.

As a final check, the CSM should answer the following questions:

- Are there potential ecological concerns?
- Is there potential for land use other than those covered by the PRGs (that is, residential and industrial)?

- Are there other likely human exposure pathways that were not considered in development of the PRGs (e.g. impact to groundwater, local fish consumption, raising beef, dairy, or other livestock)?
- Are there unusual site conditions (e.g. large areas of contamination, high fugitive dust levels, potential for indoor air contamination)?

If any of these four conditions exist, the PRG may need to be adjusted to reflect this new information. Suggested websites for the evaluation of pathways not currently addressed by Region 9 PRG's are presented in Exhibit 3-1.

**EXHIBIT 3-1
SUGGESTED WEBSITES FOR EVALUATING EXPOSURE
PATHWAYS NOT CURRENTLY ADDRESSED BY REGION 9 PRGs**

EXPOSURE PATHWAY	WEBSITE
Migration of contaminants to an underlying potable aquifer	EPA Soil Screening Guidance: http://www.epa.gov/superfund/resources/soil/index.htm California Water Board Guidance: http://www.swrcb.ca.gov/rwqcb2/rbsl.htm
Ingestion via plant uptake	EPA Soil Screening Guidance: http://www.epa.gov/superfund/resources/soil/index.htm EPA Fertilizer Risk Assessment: http://www.epa.gov/epaoswer/hazwaste/recycle/fertiliz/risk/
Ingestion via meat, dairy products, human milk	EPA Protocol for Combustion Facilities: http://www.epa.gov/epaoswer/hazwaste/combust/riskvol.htm#volume1 California "Hot Spots" Risk Guidelines: http://www.oehha.ca.gov/air/hot_spots/HRSguide.html
Inhalation of volatiles that have migrated into basements or other enclosed spaces.	EPA's Version of Johnson & Ettinger Model: http://www.epa.gov/oerrpage/superfund/programs/risk/airmodel/johnson_ettinger.htm
Ecological pathways	EPA Ecological Soil Screening Guidance: http://www.epa.gov/superfund/programs/risk/ecorisk/ecossl.htm NOAA Sediment Screening Table: http://response.restoration.noaa.gov/cpr/sediment/squirt/squirt.html

3.2 Background Levels Evaluation

A necessary step in determining the applicability of Region 9 risk-based PRGs is the consideration of background contaminant concentrations. There is new EPA guidance on determining background at sites. *Guidance for Characterizing Background Chemicals in Soil at Superfund Sites* (USEPA 2001c) is available on the web at: <http://www.epa.gov/superfund/programs/risk/background.pdf>.

EPA may be concerned with two types of background at sites: naturally occurring and anthropogenic. Natural background is usually limited to metals whereas anthropogenic (i.e. human-made) "background" includes both organic and inorganic contaminants. Before embarking on an extensive sampling and analysis program to determine local background concentrations in the area, one should first compile existing data on the subject. Far too often there is pertinent information in the literature that gets ignored, resulting in needless expenditures of time and money.

Generally EPA does not clean up below natural background. In some cases, the predictive risk-based models generate PRG levels that lie within or even below typical background. If natural background concentrations are higher than the risk-based PRGs, an adjustment of the PRG is probably needed. Exhibit 3-2 presents summary statistics for selected elements in soils that have background levels that may exceed risk-based PRGs. An illustrative example of this is naturally occurring arsenic in soils which frequently is higher than the risk-based concentration set at a one-in-one-million cancer risk (the PRG for residential soils is 0.39 mg/kg). After considering background concentrations in a local area, EPA Region 9 has at times used the non-cancer PRG (22 mg/kg) to evaluate sites recognizing that this value tends to be above background levels yet still falls within the range of soil concentrations (0.39-39 mg/kg) that equates to EPA's "acceptable" cancer risk range of $10E-6$ to $10E-4$.

Where anthropogenic "background" levels exceed PRGs and EPA has determined that a response action is necessary and feasible, EPA's goal will be to develop a comprehensive response to the widespread contamination. This will often require coordination with different authorities that have jurisdiction over the sources of contamination in the area.

EXHIBIT 3-2
BACKGROUND CONCENTRATIONS OF SELECTED ELEMENTS IN SOILS

TRACE ELEMENT	U.S. STUDY DATA ¹			CALIFORNIA DATA ²		
	Range	GeoMean	ArMean	Range	GeoMean	ArMean
Arsenic	<.1-97	5.2 mg/kg	7.2 mg/kg	0.59-11	2.75 mg/kg	3.54 mg/kg
Beryllium	<1-15	0.63 "	0.92 "	0.10-2.7	1.14 "	1.28 "
Cadmium	<1-10	--	<1	0.05-1.7	0.26	0.36
Chromium	1-2000	37	54	23-1579	76.25	122.08
Nickel	<5-700	13	19	9.0-509	35.75	56.60

¹Shacklette and Hansford, "Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States", USGS Professional Paper 1270, 1984.

²Bradford et. al, "Background Concentrations of Trace and Major Elements in California Soils", Kearney Foundation Special Report, UC-Riverside and CAL-EPA DTSC, March 1996.

3.3 Screening Sites with Multiple Pollutants

A suggested stepwise approach for PRG-screening of sites with multiple pollutants is as follows:

- Perform an extensive records search and compile existing data.
- Identify site contaminants in the PRG table. Record the PRG concentrations for various media and note whether PRG is based on cancer risk (indicated by "ca") or noncancer hazard (indicated by "nc"). Segregate cancer PRGs from non-cancer PRGs and exclude (but don't eliminate) non-risk based PRGs ("sat" or "max").
- For cancer risk estimates, take the site-specific concentration (maximum or 95 UCL) and divide by the PRG concentrations that are designated for cancer evaluation ("ca"). Multiply this ratio by 10^{-6} to estimate chemical-specific risk for a reasonable maximum exposure (RME). For multiple pollutants, simply add the risk for each chemical:

$$Risk = [(\frac{conc_x}{PRG_x}) + (\frac{conc_y}{PRG_y}) + (\frac{conc_z}{PRG_z})] \times 10^{-6}$$

- For non-cancer hazard estimates. Divide the concentration term by its respective non-cancer PRG designated as "nc" and sum the ratios for multiple contaminants. The cumulative ratio represents a non-carcinogenic hazard index (HI). A hazard index of 1 or less is generally considered "safe". A ratio greater than 1 suggests further evaluation. [Note that carcinogens may also have an associated non-cancer PRG that is not listed in the printed copy of the table sent to folks on the mailing list. To obtain these values, the user should view or download the PRG table at our website and display the appropriate sections.]

$$Hazard\ Index = [(\frac{conc_x}{PRG_x}) + (\frac{conc_y}{PRG_y}) + (\frac{conc_z}{PRG_z})]$$

For more information on screening site risks, the reader should contact EPA Region 9's Technical Support Team.

3.4 Potential Problems

As with any risk-based tool, the potential exists for misapplication. In most cases the root cause will be a lack of understanding of the intended use of Region 9 PRGs. In order to prevent misuse of PRGs, the following should be avoided:

- Applying PRGs to a site without adequately developing a conceptual site model that identifies relevant exposure pathways and exposure scenarios,
- Not considering background concentrations when choosing PRGs as cleanup goals,
- Use of PRGs as cleanup levels without the nine-criteria analysis specified in the

National Contingency Plan (or, comparable analysis for programs outside of Superfund),

- Use of PRGs as cleanup levels without verifying numbers with a toxicologist or regional risk assessor,

Use of antiquated PRG tables that have been superseded by more recent publications,

- Not considering the effects of additivity when screening multiple chemicals, and
- Adjusting PRGs upward by factors of 10 or 100 without consulting a toxicologist or regional risk assessor.

4.0 TECHNICAL SUPPORT DOCUMENTATION

Region 9 PRGs consider human exposure hazards to chemicals from contact with contaminated soils, air, and water. The emphasis of the PRG equations and technical discussion are aimed at developing screening criteria for soils, since this is an area where few standards exist. For air and water, additional reference concentrations or standards are available for many chemicals (e.g. MCLs, non-zero MCLGs, AWQC, and NAAQS) and consequently the discussion of these media are brief.

4.1 Soils - Direct Ingestion

Calculation of risk-based PRGs for direct ingestion of soil is based on methods presented in RAGS HHEM, Part B (USEPA 1991a) and *Soil Screening Guidance* (USEPA 1996a,b, USEPA 2001a). Briefly, these methods backcalculate a soil concentration level from a target risk (for carcinogens) or hazard quotient (for noncarcinogens).

Residential Soil PRGs

A number of studies have shown that inadvertent ingestion of soil is common among children 6 years old and younger (Calabrese et al. 1989, Davis et al. 1990, Van Wijnen et al. 1990). To take into account the higher soil intake rate for children, two different approaches are used to estimate PRGs, depending on whether the adverse health effect is cancer or some effect other than cancer.

For carcinogens, the method for calculating PRGs uses an age-adjusted soil ingestion factor that takes into account the difference in daily soil ingestion rates, body weights, and exposure duration for children from 1 to 6 years old and others from 7 to 31 years old. This health-protective approach is chosen to take into account the higher daily rates of soil ingestion in children as well as the longer duration of exposure that is anticipated for a long-term resident. For more on this method, see USEPA RAGs Part B (1991a).

For noncarcinogenic concerns, the more protective method of calculating a soil PRG is to evaluate childhood exposures separately from adult exposures. In other words, an age-adjustment factor is not applied as was done for carcinogens. This approach is considered conservative because it combines the higher 6-year exposure for children with chronic toxicity criteria. In their analysis of the method, the Science Advisory Board (SAB) indicated that, for most chemicals, the approach may be overly

protective. However, they noted that there are specific instances when the chronic RfD may be based on endpoints of toxicity that are specific to children (e.g. fluoride and nitrates) or when the dose-response is steep (i.e., the dosage difference between the no-observed-adverse-effects level [NOAEL] and an adverse effects level is small). Thus, for the purposes of screening, EPA Region 9 has adopted this approach for calculating soil PRGs for noncarcinogenic health concerns.

Industrial Soil PRGs

In the new *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (Supplemental SSL Guidance, EPA 2001a), two different soil ingestion rates are assumed for non-construction workers: 100 mg/day is assumed for outdoor workers whereas 50 mg/day is assumed for indoor workers. The default value of 100 mg/day for outdoor workers is also recommended by EPA's Technical Review Workgroup for Lead (TRW), and it reflects increased exposures to soils for outdoor workers relative to their indoor counterparts. For more on this, please see the Supplemental SSL Guidance available at the following website:

<http://www.epa.gov/superfund/resources/soil/index.htm>

Because the Region 9 PRGs are generic and intended for screening sites early in the investigation process (often before site-specific information is available), we have chosen to use the 100 mg/day soil ingestion (i.e. outdoor worker) assumption to calculate industrial soil PRGs. Please note that previous issues of the Region 9 PRGs table assumed 50 mg/day soil ingestion rate for workers. This change in soil ingestion rates is reflected in a somewhat lower (more stringent) industrial soils PRG for many contaminants. The appropriateness of this assumption for a particular site may be evaluated when additional information becomes available regarding site conditions or site development.

4.2 Soils - Vapor and Particulate Inhalation

Agency toxicity criteria indicate that risks from exposure to some chemicals via inhalation far outweigh the risk via ingestion; therefore soil PRGs have been designed to address this pathway as well. The models used to calculate PRGs for inhalation of volatiles/particulates are updates of risk assessment methods presented in RAGS Part B (USEPA 1991a) and are identical to the *Soil Screening Guidance: User's Guide and Technical Background Document* (USEPA 1996a,b).

It should be noted that the soil-to-air pathway that is evaluated in the PRGs calculations is based on direct inhalation exposures that result from the volatilization or particulate emissions of chemicals from soil to outdoor air. The soil PRG calculations currently do not evaluate potential for volatile contaminants in soil to migrate indoors. For this evaluation, a site-specific assessment is required because the applicable model, the Johnson and Ettinger model, is extremely sensitive to a number of model parameters that do not lend themselves to standardization on a national basis. For more information on the indoor air model and/or to download a copy, please go to:

http://www.epa.gov/oerrpage/superfund/programs/risk/airmodel/johnson_ettinger.htm

To address the soil-to-outdoor air pathways, the PRG calculations incorporate volatilization factors (VF_s) for volatile contaminants and particulate emission factors (PEF) for nonvolatile contaminants. These factors relate soil contaminant concentrations to air contaminant concentrations that may be inhaled on-site. The VF_s and PEF equations can be broken into two separate models: an emission model to estimate emissions of the contaminant from the soil and a dispersion model to simulate the dispersion of the contaminant in the atmosphere.

The box model in RAGS Part B has been replaced with a dispersion term (Q/C) derived from a modeling exercise using meteorological data from 29 locations across the United States because the box model may not be applicable to a broad range of site types and meteorology and does not utilize state-of-the-art techniques developed for regulatory dispersion modeling. The dispersion model for both volatiles and particulates is the AREA-ST, an updated version of the Office of Air Quality Planning and Standards, Industrial Source Complex Model, ISC2. However, different Q/C terms are used in the VF and PEF equations. Los Angeles was selected as the 90th percentile data set for volatiles and Minneapolis was selected as the 90th percentile data set for fugitive dusts (USEPA 1996a,b). A default source size of 0.5 acres was chosen for the PRG calculations. This is consistent with the default exposure area over which Region 9 typically averages contaminant concentrations in soils. If unusual site conditions exist such that the area source is substantially larger than the default source size assumed here, an alternative Q/C could be applied (see USEPA 1996a,b).

Volatilization Factor for Soils

Volatile chemicals, defined as those chemicals having a Henry's Law constant greater than 10^{-5} (atm-m³/mol) and a molecular weight less than 200 g/mole, were screened for inhalation exposures using a volatilization factor for soils (VF_s). Please note that VF_s 's are available at our website.

The emission terms used in the VF_s are chemical-specific and were calculated from physical-chemical information obtained from several sources. The priority of these sources were as follows: *Soil Screening Guidance* (USEPA 1996a,b), *Superfund Chemical Data Matrix* (USEPA 1996c), *Fate and Exposure Data* (Howard 1991), *Subsurface Contamination Reference Guide* (EPA 1990a), and *Superfund Exposure Assessment Manual* (SEAM, EPA 1988). When there was a choice between a measured or a modeled value (e.g. Koc), we went with modeled values. In those cases where Diffusivity Coefficients (Di) were not provided in existing literature, Di 's were calculated using Fuller's Method described in SEAM. A surrogate term was required for some chemicals that lacked physico-chemical information. In these cases, a proxy chemical of similar structure was used that may over- or under-estimate the PRG for soils.

Equation 4-9 forms the basis for deriving generic soil PRGs for the inhalation pathway. The following parameters in the standardized equation can be replaced with specific site data to develop a simple site-specific PRG

- Source area
- Average soil moisture content
- Average fraction organic carbon content
- Dry soil bulk density

The basic principle of the VF_s model (Henry's law) is applicable only if the soil contaminant concentration is at or below soil saturation "sat". Above the soil saturation limit, the model cannot predict an accurate VF-based PRG. How these particular cases are handled, depends on whether the contaminant is liquid or solid at ambient soil temperatures (see Section 4.5).

Particulate Emission Factor for Soils

Inhalation of chemicals adsorbed to respirable particles (PM₁₀) were assessed using a default PEF equal to $1.316 \times 10^9 \text{ m}^3/\text{kg}$ that relates the contaminant concentration in soil with the concentration of respirable particles in the air due to fugitive dust emissions from contaminated soils. The generic PEF was derived using default values in Equation 4-11, which corresponds to a receptor point concentration of approximately $0.76 \text{ ug}/\text{m}^3$. The relationship is derived by Cowherd (1985) for a rapid assessment procedure applicable to a typical hazardous waste site where the surface contamination provides a relatively continuous and constant potential for emission over an extended period of time (e.g. years). This represents an annual average emission rate based on wind erosion that should be compared with chronic health criteria; it is not appropriate for evaluating the potential for more acute exposures.

The impact of the PEF on the resultant PRG concentration (that combines soil exposure pathways for ingestion, skin contact, and inhalation) can be assessed by accessing the Region 9 PRG website and viewing the pathway-specific soil concentrations. Equation 4-11 forms the basis for deriving a generic PEF for the inhalation pathway. For more details regarding specific parameters used in the PEF model, the reader is referred to *Soil Screening Guidance: Technical Background Document* (USEPA 1996a).

Note: the generic PEF evaluates windborne emissions and does not consider dust emissions from traffic or other forms of mechanical disturbance that could lead to greater emissions than assumed here.

4.3 Soils - Dermal Exposure

Dermal Contact Assumptions

Exposure factors for dermal contact with soil are based on recommendations in "Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment) Interim Guidance" (USEPA 2001b). Recommended RME (reasonable maximum exposure) defaults for adult workers' skin surface areas ($3300 \text{ cm}^2/\text{day}$) and soil adherence factors ($0.2 \text{ mg}/\text{cm}^2$) now differ from the defaults recommended for adult residents ($5700 \text{ cm}^2/\text{day}$, $0.07 \text{ mg}/\text{cm}^2$) as noted in Exhibit 4-1. This is due to differences in the range of activities experienced by workers versus residents.

Dermal Absorption

Chemical-specific skin absorption values recommended by the Superfund Dermal Workgroup were applied when available. Chemical-specific values are included for the following chemicals: arsenic, cadmium, chlordane, 2,4-D, DDT, lindane, TCDD, PAHs, PCBs, and pentachlorophenols.

The "Supplemental Guidance for Dermal Risk Assessment" (USEPA 2001b) recommends a default dermal absorption factor for semivolatile organic compounds of 10% as a screening method for the majority of SVOCs without dermal absorption factors. Default dermal absorption values for other chemicals (VOCs and inorganics) are not recommended in this new guidance. Therefore, the assumption of 1% for inorganics and 10% for volatiles is no longer included in the Region 9 PRG table. This change has minimal impact on the final risk-based calculations because human exposure to VOCs and inorganics in soils is generally driven by other pathways of exposure.

4.4 Soils - Migration to Groundwater

The methodology for calculating SSLs for the migration to groundwater was developed to identify chemical concentrations in soil that have the potential to contaminate groundwater. Migration of contaminants from soil to groundwater can be envisioned as a two-stage process: (1) release of contaminant in soil leachate and (2) transport of the contaminant through the underlying soil and aquifer to a receptor well. The SSL methodology considers both of these fate and transport mechanisms.

SSLs are backcalculated from acceptable ground water concentrations (i.e. nonzero MCLGs, MCLs, or risk-based PRGs). First, the acceptable groundwater concentration is multiplied by a dilution factor to obtain a target leachate concentration. For example, if the dilution factor is 10 and the acceptable ground water concentration is 0.05 mg/L, the target soil leachate concentration would be 0.5 mg/L. The partition equation (presented in the *Soil Screening Guidance* document) is then used to calculate the total soil concentration (i.e. SSL) corresponding to this soil leachate concentration.

The SSL methodology was designed for use during the early stages of a site evaluation when information about subsurface conditions may be limited. Because of this constraint, the methodology is based on conservative, simplifying assumptions about the release and transport of contaminants in the subsurface. For more on SSLs, and how to calculate site-specific SSLs versus generic SSLs presented in the PRG table, the reader is referred to the *Soil Screening Guidance* document (USEPA 1996a,b).

4.5 Soil Saturation Limit

The soil saturation concentration "sat" corresponds to the contaminant concentration in soil at which the absorptive limits of the soil particles, the solubility limits of the soil pore water, and saturation of soil pore air have been reached. Above this concentration, the soil contaminant may be present in free phase, i.e., nonaqueous phase liquids (NAPLs) for contaminants that are liquid at ambient soil temperatures and pure solid phases for compounds that are solid at ambient soil temperatures.

Equation 4-10 is used to calculate "sat" for each volatile contaminant. As an update to RAGS HHEM, Part B (USEPA 1991a), this equation takes into account the amount of contaminant that is in the vapor phase in soil in addition to the amount dissolved in the soil's pore water and sorbed to soil particles.

Chemical-specific "sat" concentrations must be compared with each VF-based PRG because a basic principle of the PRG volatilization model is not applicable when free-phase contaminants are present. How these cases are handled depends on whether the contaminant is liquid or solid at ambient

temperatures. Liquid contaminant that have a VF-based PRG that exceeds the "sat" concentration are set equal to "sat" whereas for solids (e.g., PAHs), soil screening decisions are based on the appropriate PRGs for other pathways of concern at the site (e.g., ingestion).

4.6 Tap Water - Ingestion and Inhalation

Calculation of PRGs for ingestion and inhalation of contaminants in domestic water is based on the methodology presented in RAGS HHEM, Part B (USEPA 1991a). Ingestion of drinking water is an appropriate pathway for all chemicals. For the purposes of this guidance, however, inhalation of volatile chemicals from water is considered routinely only for chemicals with a Henry's Law constant of 1×10^{-5} atm-m³/mole or greater and with a molecular weight of less than 200 g/mole.

For volatile chemicals, an upperbound volatilization constant (VF_w) is used that is based on all uses of household water (e.g. showering, laundering, and dish washing). Certain assumptions were made. For example, it is assumed that the volume of water used in a residence for a family of four is 720 L/day, the volume of the dwelling is 150,000 L and the air exchange rate is 0.25 air changes/hour (Andelman in RAGS Part B). Furthermore, it is assumed that the average transfer efficiency weighted by water use is 50 percent (i.e. half of the concentration of each chemical in water will be transferred into air by all water uses). Note: the range of transfer efficiencies extends from 30% for toilets to 90% for dishwashers.

4.7 Default Exposure Factors

Default exposure factors were obtained primarily from RAGS Supplemental Guidance Standard Default *Exposure Factors* (OSWER Directive, 9285.6-03) dated March 25, 1991 and more recent information from U.S. EPA's Office of Solid Waste and Emergency Response, U.S. EPA's Office of Research and Development, and California EPA's Department of Toxic Substances Control (see Exhibit 4-1).

Because contact rates may be different for children and adults, carcinogenic risks during the first 30 years of life were calculated using age-adjusted factors ("adj"). Use of age-adjusted factors are especially important for soil ingestion exposures, which are higher during childhood and decrease with age. However, for purposes of combining exposures across pathways, additional age-adjusted factors are used for inhalation and dermal exposures. These factors approximate the integrated exposure from birth until age 30 combining contact rates, body weights, and exposure durations for two age groups - small children and adults. Age-adjusted factors were obtained from RAGS PART B or developed by analogy (see derivations next page).

For soils only, noncarcinogenic contaminants are evaluated in children separately from adults. No age-adjustment factor is used in this case. The focus on children is considered protective of the higher daily intake rates of soil by children and their lower body weight. For maintaining consistency when evaluating soils, dermal and inhalation exposures are also based on childhood contact rates.

- (1) ingestion([mg-yr]/[kg-d]):

$$IFS_{adj} = \frac{ED_c \times IRS_c}{BW_c} + \frac{(ED_r - ED_c) \times IRS_a}{BW_a}$$

- (2) skin contact([mg-yr]/[kg-d]):

$$SFS_{adj} = \frac{ED_c \times AF \times SA_c}{BW_c} + \frac{(ED_r - ED_c) \times AF \times SA_a}{BW_a}$$

- (3) inhalation ([m³-yr]/[kg-d]):

$$InhF_{adj} = \frac{ED_c \times IRA_c}{BW_c} + \frac{(ED_r - ED_c) \times IRA_a}{BW_a}$$

EXHIBIT 4-1 STANDARD DEFAULT FACTORS

<u>Symbol</u>	<u>Definition (units)</u>	<u>Default</u>	<u>Reference</u>
CSFo	Cancer slope factor oral (mg/kg-d)-1	--	IRIS, HEAST, or NCEA
CSFi	Cancer slope factor inhaled (mg/kg-d)-1	--	IRIS, HEAST, or NCEA
RfDo	Reference dose oral (mg/kg-d)	--	IRIS, HEAST, or NCEA
RfDi	Reference dose inhaled (mg/kg-d)	--	IRIS, HEAST, or NCEA
TR	Target cancer risk	10 ⁻⁶	--
THQ	Target hazard quotient	1	--
BWa	Body weight, adult (kg)	70	RAGS (Part A), EPA 1989 (EPA/540/1-89/002)
BWc	Body weight, child (kg)	15	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
ATc	Averaging time - carcinogens (days)	25550	RAGS(Part A), EPA 1989 (EPA/540/1-89/002)
ATn	Averaging time - noncarcinogens (days)	ED*365	
SAA	Exposed surface area for soil/dust (cm ² /day)		Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
	– adult resident	5700	
	– adult worker	3300	
SAC	Exposed surface area, child in soil (cm ² /day)	2800	Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
AFa	Adherence factor, soils (mg/cm ²)		Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
	– adult resident	0.07	
	– adult worker	0.2	
AFc	Adherence factor, child (mg/cm ²)	0.2	Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
ABS	Skin absorption defaults (unitless):		
	– semi-volatile organics	0.1	Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
	– volatile organics	--	Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
	– inorganics	--	Dermal Assessment, EPA 2000 (EPA/540/R-99/005)
IRAA	Inhalation rate - adult (m ³ /day)	20	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
IRAc	Inhalation rate - child (m ³ /day)	10	Exposure Factors, EPA 1997 (EPA/600/P-95/002Fa)
IRWA	Drinking water ingestion - adult (L/day)	2	RAGS(Part A), EPA 1989 (EPA/540/1-89/002)
IRWc	Drinking water ingestion - child (L/day)	1	PEA, Cal-EPA (DTSC, 1994)
IRSa	Soil ingestion - adult (mg/day)	100	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
IRSc	Soil ingestion - child (mg/day)	200	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
IRSo	Soil ingestion - occupational (mg/day)	100	Soil Screening Guidance (EPA 2001a)
EFr	Exposure frequency - residential (d/y)	350	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
EFo	Exposure frequency - occupational (d/y)	250	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
EDr	Exposure duration - residential (years)	30 ^a	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
EDc	Exposure duration - child (years)	6	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
EDo	Exposure duration - occupational (years)	25	Exposure Factors, EPA 1991 (OSWER No. 9285.6-03)
	Age-adjusted factors for carcinogens:		
IFSadj	Ingestion factor, soils ((mg-yr)/(kg-d))	114	RAGS(Part B), EPA 1991 (OSWER No. 9285.7-01B)
SFSadj	Dermal factor, soils ((mg-yr)/(kg-d))	361	By analogy to RAGS (Part B)
InhFadj	Inhalation factor, air ((m ³ -yr)/(kg-d))	11	By analogy to RAGS (Part B)
IFWadj	Ingestion factor, water ((L-yr)/(kg-d))	1.1	By analogy to RAGS (Part B)
VFW	Volatilization factor for water (L/m ³)	0.5	RAGS(Part B), EPA 1991 (OSWER No. 9285.7-01B)
PEF	Particulate emission factor (m ³ /kg)	See below	Soil Screening Guidance (EPA 1996a,b)
VF _s	Volatilization factor for soil (m ³ /kg)	See below	Soil Screening Guidance (EPA 1996a,b)
sat	Soil saturation concentration (mg/kg)	See below	Soil Screening Guidance (EPA 1996a,b)

Footnote:

^aExposure duration for lifetime residents is assumed to be 30 years total. For carcinogens, exposures are combined for children (6 years) and adults (24 years).

4.8 Standardized Equations

The equations used to calculate the PRGs for carcinogenic and noncarcinogenic contaminants are presented in Equations 4-1 through 4-8. The PRG equations update RAGS Part B equations. The methodology backcalculates a soil, air, or water concentration level from a target risk (for carcinogens) or hazard quotient (for noncarcinogens). For completeness, the soil equations combine risks from ingestion, skin contact, and inhalation simultaneously. **Note: the electronic version of the table also includes pathway-specific PRGs, should the user decide against combining specific exposure pathways; or, the user wants to identify the relative contribution of each pathway to exposure.**

To calculate PRGs for volatile chemicals in soil, a chemical-specific volatilization factor is calculated per Equation 4-9. Because of its reliance on Henry's law, the VF_s model is applicable only when the contaminant concentration in soil is at or below saturation (i.e. there is no free-phase contaminant present). Soil saturation ("sat") corresponds to the contaminant concentration in soil at which the adsorptive limits of the soil particles and the solubility limits of the available soil moisture have been reached. Above this point, pure liquid-phase contaminant is expected in the soil. If the PRG calculated using VF_s was greater than the calculated sat, the PRG was set equal to sat, in accordance with *Soil Screening Guidance* (USEPA 1996 a,b). The equation for deriving sat is presented in Equation 4-10.

PRG EQUATIONS

Soil Equations: For soils, equations were based on three exposure routes (ingestion, skin contact, and inhalation).

Equation 4-1: Combined Exposures to Carcinogenic Contaminants in Residential Soil

$$C(\text{mg/kg}) = \frac{TR \times AT_c}{EF_r \left[\left(\frac{IFS_{adj} \times CSF_o}{10^6 \text{ mg/kg}} \right) + \left(\frac{SFS_{adj} \times ABS \times CSF_o}{10^6 \text{ mg/kg}} \right) + \left(\frac{InhF_{adj} \times CSF_i}{VF_s^a} \right) \right]}$$

Equation 4-2: Combined Exposures to Noncarcinogenic Contaminants in Residential Soil

$$C(\text{mg/kg}) = \frac{THQ \times BW_c \times AT_n}{EF_r \times ED_c \left[\left(\frac{1}{RFD_o} \times \frac{IRS_c}{10^6 \text{ mg/kg}} \right) + \left(\frac{1}{RFD_o} \times \frac{SA_c \times AF \times ABS}{10^6 \text{ mg/kg}} \right) + \left(\frac{1}{RFD_i} \times \frac{IRA_c}{VF_s^a} \right) \right]}$$

Equation 4-3: Combined Exposures to Carcinogenic Contaminants in Industrial Soil

$$C(\text{mg/kg}) = \frac{TR \times BW_a \times AT_c}{EF_o \times ED_o \left[\left(\frac{IRS_o \times CSF_o}{10^6 \text{ mg/kg}} \right) + \left(\frac{SA_a \times AF \times ABS \times CSF_o}{10^6 \text{ mg/kg}} \right) + \left(\frac{IRA_a \times CSF_i}{VF_s^a} \right) \right]}$$

Footnote:

^aUse VF_s for volatile chemicals (defined as having a Henry's Law Constant [atm-m³/mol] greater than 10^{-5} and a molecular weight less than 200 grams/mol) or PEF for non-volatile chemicals.

Equation 4-4: Combined Exposures to Noncarcinogenic Contaminants in Industrial Soil

$$C(\text{mg/kg}) = \frac{THQ \times BW_a \times AT_n}{EF_o \times ED_o \left[\left(\frac{1}{RfD_o} \times \frac{IRS_o}{10^6 \text{mg/kg}} \right) + \left(\frac{1}{RfD_o} \times \frac{SA_a \times AF \times ABS}{10^6 \text{mg/kg}} \right) + \left(\frac{1}{RfD_i} \times \frac{IRA_a}{VF_a} \right) \right]}$$

Tap Water Equations:

Equation 4-5: Ingestion and Inhalation Exposures to Carcinogenic Contaminants in Water

$$C(\text{ug/L}) = \frac{TR \times AT_c \times 1000 \text{ug/mg}}{EF_r \left[(IFW_{adj} \times CSF_o) + (VF_w \times InhF_{adj} \times CSF_i) \right]}$$

Equation 4-6: Ingestion and Inhalation Exposures to Noncarcinogenic Contaminants in Water

$$C(\text{ug/L}) = \frac{THQ \times BW_a \times AT_n \times 1000 \text{ug/mg}}{EF_r \times ED_r \left[\left(\frac{IRW_a}{RfD_o} \right) + \left(\frac{VF_w \times IRA_a}{RfD_i} \right) \right]}$$

Air Equations:

Equation 4-7: Inhalation Exposures to Carcinogenic Contaminants in Air

$$C(\text{ug/m}^3) = \frac{TR \times AT_c \times 1000 \text{ug/mg}}{EF_r \times InhF_{adj} \times CSF_i}$$

Equation 4-8: Inhalation Exposures to Noncarcinogenic Contaminants in Air

$$C(\text{ug/m}^3) = \frac{THQ \times RfD_i \times BW_a \times AT_n \times 1000 \text{ug/mg}}{EF_r \times ED_r \times IRA_a}$$

Footnote:

*Use VF_i for volatile chemicals (defined as having a Henry's Law Constant [atm-m³/mol] greater than 10⁻⁵ and a molecular weight less than 200 grams/mol) or PEF for non-volatile chemicals.

SOIL-TO-AIR VOLATILIZATION FACTOR (VF_s)

Equation 4-9: Derivation of the Volatilization Factor

$$VF_s (m^3/kg) = (Q/C) \times \frac{(3.14 \times D_A \times T)^{1/2}}{(2 \times \rho_b \times D_A)} \times 10^{-4} (m^2/cm^2)$$

where:

$$D_A = \frac{[(\Theta_a^{10/3} D_i H' + \Theta_w^{10/3} D_w) / n^2]}{\rho_b K_d + \Theta_w + \Theta_a H'}$$

<u>Parameter</u>	<u>Definition (units)</u>	<u>Default</u>
VF _s	Volatilization factor (m ³ /kg)	--
D _A	Apparent diffusivity (cm ² /s)	--
Q/C	Inverse of the mean conc. at the center of a 0.5-acre square source (g ^{M2} -s per kg/m ³)	68.81
T	Exposure interval (s)	9.5 x 10 ⁸
ρ _b	Dry soil bulk density (g/cm ³)	1.5
Θ _a	Air filled soil porosity (L _{air} /L _{soil})	0.28 or n-Θ _w
n	Total soil porosity (L _{pore} /L _{soil})	0.43 or 1 - (ρ _b /ρ _s)
Θ _w	Water-filled soil porosity (L _{water} /L _{soil})	0.15
ρ _s	Soil particle density (g/cm ³)	2.65
D _i	Diffusivity in air (cm ² /s)	Chemical-specific
H	Henry's Law constant (atm-m ³ /mol)	Chemical-specific
H'	Dimensionless Henry's Law constant	Calculated from H by multiplying by 41 (USEPA 1991a)
D _w	Diffusivity in water (cm ² /s)	Chemical-specific
K _d	Soil-water partition coefficient (cm ³ /g) = K _{oc} f _{oc}	Chemical-specific
K _{oc}	Soil organic carbon-water partition coefficient (cm ³ /g)	Chemical-specific
f _{oc}	Fraction organic carbon in soil (g/g)	0.006 (0.6%)

SOIL SATURATION CONCENTRATION (sat)

Equation 4-10: Derivation of the Soil Saturation Limit

$$sat = \frac{S}{\rho_b} (K_d \rho_b + \Theta_w + H' \Theta_a)$$

<u>Parameter</u>	<u>Definition (units)</u>	<u>Default</u>
sat	Soil saturation concentration (mg/kg)	--
S	Solubility in water (mg/L-water)	Chemical-specific
ρ_b	Dry soil bulk density (kg/L)	1.5
n	Total soil porosity (L_{pore}/L_{soil})	0.43 or $1 - (\rho_b/\rho_s)$
ρ_s	Soil particle density (kg/L)	2.65
K_d	Soil-water partition coefficient (L/kg)	$K_{oc} \times f_{oc}$ (chemical-specific)
k_{oc}	Soil organic carbon/water partition coefficient (L/kg)	Chemical-specific
f_{oc}	Fraction organic carbon content of soil (g/g)	0.006 or site-specific
Θ_w	Water-filled soil porosity (L_{water}/L_{soil})	0.15
Θ_a	Air filled soil porosity (L_{air}/L_{soil})	0.28 or $n - \Theta_w$
w	Average soil moisture content (kg_{water}/kg_{soil} or L_{water}/L_{soil})	0.1
H	Henry's Law constant (atm-m ³ /mol)	Chemical-specific
H'	Dimensionless Henry's Law constant	$H \times 41$, where 41 is a units conversion factor

SOIL-TO-AIR PARTICULATE EMISSION FACTOR (PEF)

Equation 4-11: Derivation of the Particulate Emission Factor

$$PEF(m^3/kg) = Q/C \times \frac{3600s/h}{0.036 \times (1-V) \times (U_m/U_t)^3 \times F(x)}$$

<u>Parameter</u>	<u>Definition (units)</u>	<u>Default</u>
PEF	Particulate emission factor (m ³ /kg)	1.316 x 10 ⁹
Q/C	Inverse of the mean concentration at the center of a 0.5-acre-square source (g/M ² -s per kg/m ³)	90.80
V	Fraction of vegetative cover (unitless)	0.5
U _m	Mean annual windspeed (m/s)	4.69
U _t	Equivalent threshold value of windspeed at 7 m (m/s)	11.32
F(x)	Function dependent on U _m /U _t derived using Cowherd (1985) (unitless)	0.194

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APPENDIX 3

RELEVANT PORTIONS OF
*USER'S GUIDE FOR THE JOHNSON AND
ETTINGER (1991) MODEL FOR SUBSURFACE
VAPOR INTRUSION INTO BUILDINGS*

SENSITIVITY ANALYSIS OF *JOHNSON AND
ETTINGER (1991) MODEL*

INTERIM FINAL – JULY 2003
SF Bay RWQCB

**USER'S GUIDE FOR
THE JOHNSON AND ETTINGER (1991) MODEL
FOR SUBSURFACE VAPOR INTRUSION
INTO BUILDINGS
(REVISED)**

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WHAT'S NEW IN THIS VERSION!

This revised version of the User's Guide corresponds with the release of Version 2.3 of the Johnson and Ettinger (1991) model spreadsheets for estimating subsurface vapor intrusion into buildings. Several things have changed within the models since the original version was released in September, 1998. The following represent the major changes in Version 2.3.

1. The mean values of the van Genuchten (1980) soil water retention parameters and the values of the soil saturated hydraulic conductivity by U.S. Soil Conservation Service (SCS) soil textural classification have changed. The original values were from Carsel and Parrish (1988) and represented calculated values based on regression analyses. The new SCS "class average" values are from Schaap and Leij (1998) and represent empirical measurements. These values are used in the groundwater models to estimate the water-filled porosity in the capillary zone and are also used in all the models to estimate the soil vapor permeability. The effect of using the new values is mixed. Compared with using the old parameter values, the new estimated values of the capillary zone water-filled porosity and the soil vapor permeability will increase for some SCS soil types and decrease for others.
2. The values of the organic carbon partition coefficient (K_{oc}) for 2,4-dinitrophenol and benzoic acid have changed.
3. The unit risk factors (URF) for chlordane, benzene, 1,3-dichloropropene, and vinyl chloride have changed.
4. The reference concentration (RfC) for chlordane has changed and a RfC for vinyl chloride has been added.
5. References have been added in the text to the new 4-phase models (NAPL-SCREEN and NAPL-ADV) for estimating indoor vapor intrusion from residual-phase (e.g., NAPL) contamination in soils. These new models will soon be made available on the EPA Superfund Risk Assessment Website.
6. The previous version of the User's Guide explained that the old soil models were not applicable for concentrations greater than the soil saturation concentration (C_{sat}), nor were the groundwater models applicable for concentrations greater than the solubility limit in water (S). In such cases, a residual or nonaqueous-phase exists and the vapor concentration at the source of emissions (C_{source}) is at its highest value regardless of the initial concentration. In the previous version of the models, however, there were no automatic checks to see if inappropriate values had been entered. The new versions of the model (as explained in the User's Guide) contain such

automatic checks. If the user enters an initial soil concentration greater than C_{sat} or a groundwater concentration greater than S , the model will calculate the value of C_{source} at the saturation vapor concentration.

In addition to the release of the new Version 2.3 models, two new models have been added for evaluating subsurface vapor intrusion into buildings using empirical soil gas data. The new models (SG-SCREEN and SG-ADV) allow the user to input soil gas concentration and sampling depth information directly into the spreadsheets. The models will subsequently estimate the resulting steady-state indoor air concentration as well as the associated health risks.

SECTION 1

INTRODUCTION

Volatilization of contaminants located in subsurface soils or in groundwater, and the subsequent mass transport of these vapors into indoor spaces constitutes a potential inhalation exposure pathway which may need to be evaluated when preparing risk assessments. Likewise, this potential indoor inhalation exposure pathway may need evaluation when estimating a risk-based soil or groundwater concentration below which associated adverse health effects are unlikely.

Johnson and Ettinger (1991) introduced a screening-level model which incorporates both convective and diffusive mechanisms for estimating the transport of contaminant vapors emanating from either subsurface soils or groundwater into indoor spaces located directly above the source of contamination. In their article, Johnson and Ettinger reported that the results of the model were in qualitative agreement with published experimental case histories and in good qualitative and quantitative agreement with detailed three-dimensional numerical modeling of radon transport into houses (Loureiro, et al., 1990).

The Johnson and Ettinger model is a one-dimensional analytical solution to convective and diffusive vapor transport into indoor spaces and provides an estimated attenuation coefficient that relates the vapor concentration in the indoor space to the vapor concentration at the source of contamination. The model is constructed as both a steady-state solution to vapor transport (infinite or nondiminishing source) and as a quasi-steady-state solution (finite or diminishing source). Inputs to the model include chemical properties of the contaminant, saturated and unsaturated zone soil properties, and structural properties of the building.

This manual provides documentation and instructions for using the Johnson and Ettinger model as provided in the accompanying spreadsheets. The infinite source model and the finite source model are provided in both MICROSOFT EXCEL and LOTUS 1-2-3

formats for soil contamination and the infinite source model for contamination occurring below the water table. Model results (both screening and advanced) are provided as either a risk-based soil or groundwater concentration, or as an estimate of the actual incremental risks associated with a user-defined initial concentration. That is to say that the model will reverse-calculate an "acceptable" soil or groundwater concentration given a user-defined risk level (i.e., target risk level or target hazard quotient), or the model may be used to forward-calculate an incremental cancer risk or hazard quotient based on an initial soil or groundwater concentration.

The infinite source models for soil contamination and groundwater contamination should be used as first-tier screening tools. In these models, all but the most sensitive model parameters have been set equal to central tendency or upper bound values. Values for the most sensitive parameters may be user-defined.

More rigorous estimates may be obtained using site-specific data and the finite source model for soil contamination. Because the source of groundwater contamination may be located upgradient of the enclosed structure for which the indoor inhalation pathway is to be assessed, the advanced model for contaminated groundwater is based on an infinite source of contamination, however, site-specific values for all other model parameters may be user-defined.

In addition to the finite and infinite source models referred to above, two new models have been added (SG-SCREEN and SG-ADV) that allow the user to input empirical soil gas concentration and sampling depth information directly into the spreadsheets. These models will subsequently estimate the resulting steady-state indoor air concentrations and associated health risks.

Because of the paucity of empirical data available for either bench-scale or field-scale verification of the accuracy of these models, as well as for other vapor intrusion models, the user is advised to consider the variation of input parameters and to explore and quantify the impacts of assumptions on the uncertainty of model results. At a minimum, a range of results should be generated based on variation of the most sensitive model parameters (Section 6).

SECTION 2

MODEL THEORY

Chemical fate and transport within soils and between the soil column and enclosed spaces is determined by a number of physical and chemical processes. This section presents the theoretical framework on which the Johnson and Ettinger model is based, taking into account the most significant of these processes. In addition, this section also presents the theoretical basis for estimating values for some of the most sensitive model parameters when empirical field data are lacking. The fundamental theoretical development of this model was performed by Johnson and Ettinger (1991).

2.1 MODEL SETTING

Consider a contaminant vapor source (C_{source}) located some distance (L_T) below the floor of an enclosed building constructed with a basement or constructed slab-on-grade. The source of contamination is either a soil-incorporated volatile contaminant or a volatile contaminant in solution with groundwater below the top of the water table.

Figure 1 is a simplified conceptual diagram of the scenario where the source of contamination is incorporated in soil and buried some distance below the enclosed space floor. At the top boundary of contamination, molecular diffusion moves the volatilized contaminant towards the soil surface until it reaches the zone of influence of the building. Here convective air movement within the soil column transports the vapors through cracks between the foundation and the basement slab floor. This convective sweep effect is induced by a negative pressure within the structure caused by a combination of wind effects and stack effects due to building heating and mechanical ventilation.

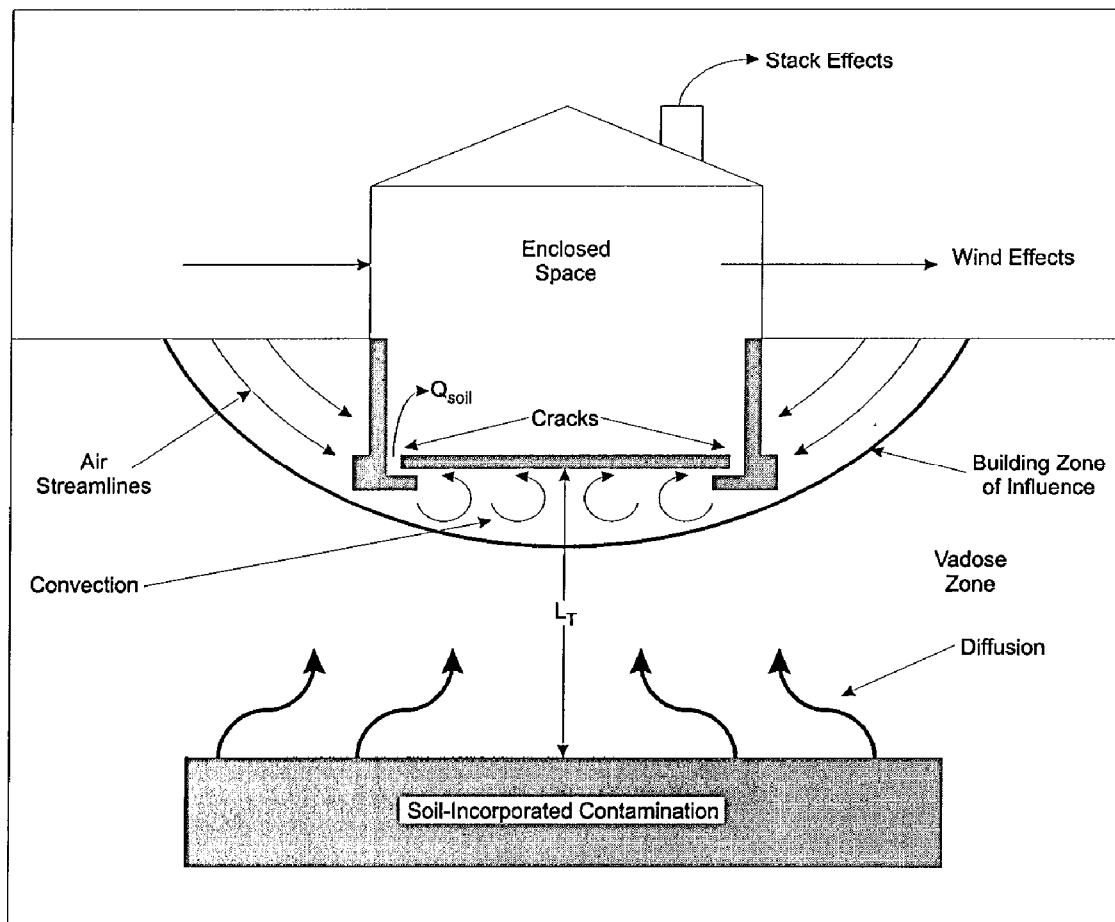


Figure 1. Conceptual Diagram of Soil Contamination

Figure 2 illustrates the scenario where the source of contamination is below the top of the water table. Here the contaminant must diffuse through a capillary zone immediately above the water table and through the subsequent unsaturated or vadose zone before convection transports the vapors into the structure.

The rate of soil gas entry (Q_{soil}) is a function solely of convection, however, the vapor concentration entering the structure may be limited by either convection or diffusion depending upon the magnitude of the source-building separation (L_T).

2.2 VAPOR CONCENTRATION AT THE SOURCE OF CONTAMINATION

With a general concept of the problem under consideration, the solution begins with an estimate of the vapor concentration at the source of contamination.

In the case of soil contamination, the initial concentration (C_R) does not contain a residual-phase (e.g., nonaqueous-phase liquid or solid); and in the case of contaminated groundwater, the initial contaminant concentration (C_W) is less than the aqueous solubility limit (i.e., in solution with water).

Given these initial conditions, C_{source} for soil contamination may be estimated from Johnson et al. (1990) as:

$$C_{\text{source}} = \frac{H'_{TS} C_R \rho_b}{\theta_w + K_d \rho_b + H'_{TS} \theta_a} \quad (1)$$

where C_{source} = Vapor concentration at the source of contamination, g/cm³-v

H'_{TS} = Henry's law constant at the system (soil) temperature,
dimensionless

C_R = Initial soil concentration, g/g

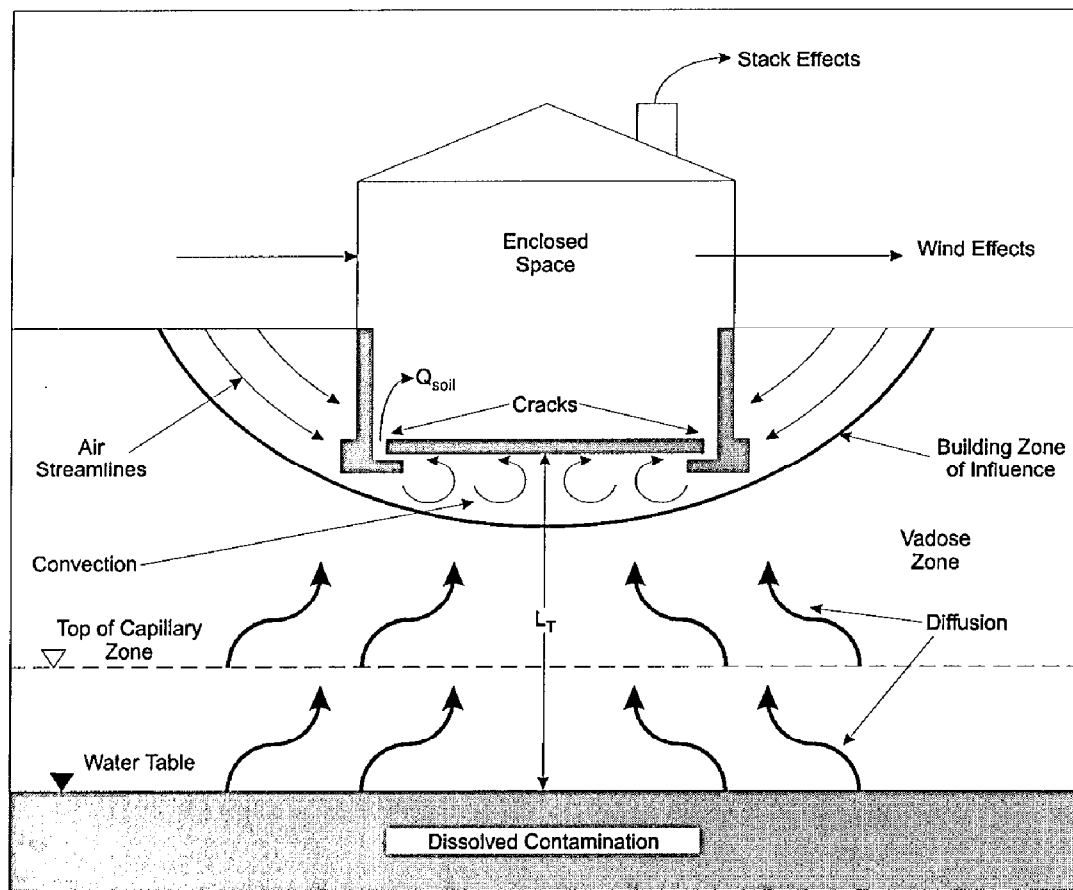


Figure 2. Conceptual Diagram of Groundwater Contamination

- ρ_b = Soil dry bulk density, g/cm³
 θ_w = Soil water-filled porosity, cm³/cm³
 K_d = Soil-water partition coefficient, cm³/g (= $K_{oc} \times f_{oc}$)
 θ_a = Soil air-filled porosity, cm³/cm³
 K_{oc} = Soil organic carbon partition coefficient, cm³/g
 f_{oc} = Soil organic carbon weight fraction.

If the initial soil concentration includes a residual phase, the user is referred to the NAPL-SCREEN or NAPL-ADV models and the accompanying user's guide. These models estimate indoor air concentrations and associated risks for up to 10 user-defined contaminants that comprise a residual phase mixture in soils.

C_{source} for groundwater contamination is estimated assuming that the vapor and aqueous-phases are in local equilibrium according to Henry's law such that:

$$C_{source} = H'_{TS} C_w \quad (2)$$

- where
- C_{source} = Vapor concentration at the source of contamination, g/cm³-v
 - H'_{TS} = Henry's law constant at the system (groundwater) temperature, dimensionless
 - C_w = Groundwater concentration, g/cm³-w.

The dimensionless form of the Henry's law constant at the system temperature (i.e., at the average soil/groundwater temperature) may be estimated using the Clapeyron equation by:

$$H'_{TS} = \frac{\exp\left[-\frac{\Delta H_{v,TS}}{R_c} \left(\frac{1}{T_s} - \frac{1}{T_R}\right)\right] H_R}{RT_s} \quad (3)$$

where H'_{TS} = Henry's law constant at the system temperature, dimensionless

$\Delta H_{v,TS}$ = Enthalpy of vaporization at the system temperature, cal/mol

T_S = System temperature, °K

T_R = Henry's law constant reference temperature, °K

H_R = Henry's law constant at the reference temperature, atm-m³/mol

R_C = Gas constant (= 1.9872 cal/mol - °K)

R = Gas constant (= 8.205 E-05 atm-m³/mol-°K).

The enthalpy of vaporization at the system temperature can be calculated from Lyman et al. (1990) as:

$$\Delta H_{v,TS} = \Delta H_{v,b} \left[\frac{(1 - T_S/T_C)}{(1 - T_B/T_C)} \right]^n \quad (4)$$

where $\Delta H_{v,TS}$ = Enthalpy of vaporization at the system temperature, cal/mol

$\Delta H_{v,b}$ = Enthalpy of vaporization at the normal boiling point, cal/mol

T_S = System temperature, °K

T_C = Critical temperature, °K

T_B = Normal boiling point, °K

n = Constant, unitless.

Table 1 gives the value of n as a function of the ratio T_B/T_C .

TABLE 1. VALUES OF EXPONENT n AS A FUNCTION OF T_B/T_C

T_B/T_C	n
< 0.57	0.30
0.57 - 0.71	$0.74 (T_B/T_C) - 0.116$
> 0.71	0.41

Chemical properties are included in the accompanying spreadsheets for the 93 volatile chemicals listed in the U.S. Environmental Protection Agency (EPA) Soil Screening Guidance (EPA 1996a and b). See Appendix C for the complete list of references by chemical.

2.3 DIFFUSION THROUGH THE CAPILLARY ZONE

Directly above the water table, a saturated capillary zone exists whereby groundwater is held within the soil pores at less than atmospheric pressure (Freeze and Cherry, 1979). Between drainage and wetting conditions, the saturated water content varies but is always less than the fully saturated water content which is equal to the soil total porosity. This is the result of air entrapment in the pores during the wetting process (Gillham, 1984). Upon rewetting, the air content of the capillary zone will be higher than after main drainage. Therefore, the air content will vary as a function of groundwater recharge and discharge. At the saturated water content, Freijer (1994) found that the relative vapor-phase diffusion coefficient was almost zero implying that all remaining air-filled soil pores are disconnected and thus blocked for gas diffusion. As the air-filled porosity increased, however, the relative diffusion coefficient indicated the presence of connected air-filled pores which corresponded to the air-entry pressure head. The air-entry pressure head corresponds with the top of the saturated capillary zone. Therefore, to allow for the calculation of the effective diffusion coefficient by lumping the gas-phase and aqueous-phase together, the water-filled soil porosity in the capillary zone ($\theta_{w,cz}$) is calculated at the air-entry pressure head (h) according to the procedures of Waitz et al.

(1996) and the van Genuchten equation (van Genuchten, 1980) for the water retention curve:

$$\theta_{w,cz} = \theta_r + \frac{\theta_s - \theta_r}{[1 + (\alpha h)^N]^M} \quad (5)$$

- where
- $\theta_{w,cz}$ = Water-filled porosity in the capillary zone, cm³/cm³
 - θ_r = Residual soil water content, cm³/cm³
 - θ_s = Saturated soil water content, cm³/cm³
 - α = Point of inflection in the water retention curve where d θ_w /dh is maximal, cm⁻¹
 - h = Air-entry pressure head, cm (= 1/ α and assumed to be positive)
 - N = van Genuchten curve shape parameter, dimensionless
 - M = 1 - (1/ N).

With a calculated value of $\theta_{w,cz}$ within the capillary zone at the air-entry pressure head, the air-filled porosity within the capillary zone ($\theta_{a,cz}$) corresponding to the minimum value at which gas diffusion is relevant is calculated as the total porosity (n) minus $\theta_{w,cz}$.

Schaap and Leij (1998) computed the Soil Conservation Service (SCS) class average values of the van Genuchten soil water retention curve parameters for each of the 12 SCS soil textural classifications. Table 2 provides the class average values for each of the SCS soil types. These data replace the mean values developed by Carsel and Parrish (1988) included in the previous EPA versions of the Johnson and Ettinger models. With the class average values presented in Table 2, a general estimate can be made of the values of $\theta_{w,cz}$ and $\theta_{a,cz}$ for each soil textural classification.

The total concentration effective diffusion coefficient across the capillary zone (D_{cz}^{eff}) may then be calculated using the Millington and Quirk (1961) model as:

TABLE 2. CLASS AVERAGE VALUES OF THE VAN GENUCHTEN SOIL WATER RETENTION PARAMETERS FOR THE 12 SCS SOIL TEXTURAL CLASSIFICATIONS

Soil texture (USDA)	Saturated water content, θ_s	Residual water Content, θ_r	van Genuchten parameters			Number of samples
			α (1/cm)	N	M	
Clay	0.459	0.098	0.01496	1.253	0.2019	84
Clay loam	0.442	0.079	0.01581	1.416	0.2938	140
Loam	0.399	0.061	0.01112	1.472	0.3207	242
Loamy sand	0.390	0.049	0.03475	1.746	0.4273	201
Silt	0.489	0.050	0.00658	1.679	0.4044	6
Silt loam	0.439	0.065	0.00506	1.663	0.3987	330
Silty clay	0.481	0.111	0.01622	1.321	0.2430	28
Silty clay loam	0.482	0.090	0.00839	1.521	0.3425	172
Sand	0.375	0.053	0.03524	3.177	0.6852	308
Sandy clay	0.385	0.117	0.03342	1.208	0.1722	11
Sandy clay loam	0.384	0.063	0.02109	1.330	0.2481	87
Sandy loam	0.387	0.039	0.02667	1.449	0.3099	476

$$D_{cz}^{eff} = D_a (\theta_{a,cz}^{3.33} / n_{cz}^2) + (D_w / H'_{TS}) (\theta_{w,cz}^{3.33} / n_{cz}^2) \quad (6)$$

where D_{cz}^{eff} = Effective diffusion coefficient across the capillary zone, cm^2/s

D_a = Diffusivity in air, cm^2/s

$\theta_{a,cz}$ = Soil air-filled porosity in the capillary zone, cm^3/cm^3

n_{cz} = Soil total porosity in the capillary zone, cm^3/cm^3

D_w = Diffusivity in water, cm^2/s

H'_{TS} = Henry's law constant at the system temperature, dimensionless

$\theta_{w,cz}$ = Soil water-filled porosity in the capillary zone, cm^3/cm^3 .

According to Fick's law of diffusion, the rate of mass transfer across the capillary zone can be approximated by the expression:

$$E = A(C_{\text{source}} - C_{g0})D_{cz}^{\text{eff}} / L_{cz} \quad (7)$$

where	E	= Rate of mass transfer, g/s
	A	= Cross-sectional area through which vapors pass, cm^2
	C_{source}	= Vapor concentration within the capillary zone, g/cm^3 -v
	C_{g0}	= A known vapor concentration at the top of the capillary zone, g/cm^3 -v (C_{g0} is assumed to be zero as diffusion proceeds upward)
	D_{cz}^{eff}	= Effective diffusion coefficient across the capillary zone, cm^2/s
	L_{cz}	= Thickness of capillary zone, cm.

The value of C_{source} is calculated using Equation 2; the value of A is assumed to be one cm^2 ; and the value of D_{cz}^{eff} is calculated by Equation 6. What remains is a way to estimate a value for L_{cz} .

Lohman (1972) and Fetter (1994) estimated the rise of the capillary zone above the water table using the phenomenon of capillarity such that water molecules are subject to an upward attractive force due to surface tension at the air-water interface and the molecular attraction of the liquid and solid phases. The rise of the capillary zone can thus be estimated using the equation for the height of capillary rise in a bundle of tubes of various diameters equivalent to the diameters between varying soil grain sizes. Fetter (1994) estimated the mean rise of the capillary zone as:

$$L_{cz} = \frac{2 \alpha \cos \lambda}{\rho_w g R} \quad (8)$$

where L_{cz} = Mean rise of the capillary zone, cm
 α = Surface tension of water, g/s (= 73)
 λ = Angle of the water meniscus with the capillary tube, degrees
 (assumed to be zero)
 ρ_w = Density of water, g/cm³ (= 0.999)
 g = Acceleration due to gravity, cm/s² (= 980)
 R = Mean interparticle pore radius, cm

and,

$$R = 0.2D \quad (9)$$

where R = Mean interparticle pore radius, cm
 D = Mean particle diameter, cm.

Assuming that the default values of the parameters given in Equation 8 are for groundwater between 5° and 25°C, Equation 8 reduces to:

$$L_{cz} = \frac{0.15}{R} \quad (10)$$

Nielson and Rogers (1990) estimated the arithmetic mean particle diameter for each of the 12 SCS soil textural classifications at the mathematical centroid calculated from its classification area (Figure 3). Table 3 shows the centroid compositions and mean particle sizes of the 12 SCS soil textural classes.

Given the mean particle diameter data in Table 3, the mean thickness of the capillary zone may then be estimated using Equations 9 and 10.

2.4 DIFFUSION THROUGH THE UNSATURATED ZONE

The effective diffusion coefficient within the unsaturated zone may also be estimated using the same form as Equation 6:

$$D_i^{eff} = D_a (\theta_{a,i}^{3.33} / n_i^2) + (D_w / H'_{TS}) (\theta_{w,i}^{3.33} / n_i^2) \quad (11)$$

where

- D_i^{eff} = Effective diffusion coefficient across soil layer i, cm²/s
- D_a = Diffusivity in air, cm²/s
- $\theta_{a,i}$ = Soil air-filled porosity of layer i, cm³/cm³
- n_i = Soil total porosity of layer i, cm³/cm³
- D_w = Diffusivity in water, cm²/s
- $\theta_{w,i}$ = Soil water-filled porosity of layer i, cm³/cm³
- H'_{TS} = Henry's law constant at the system temperature, dimensionless.

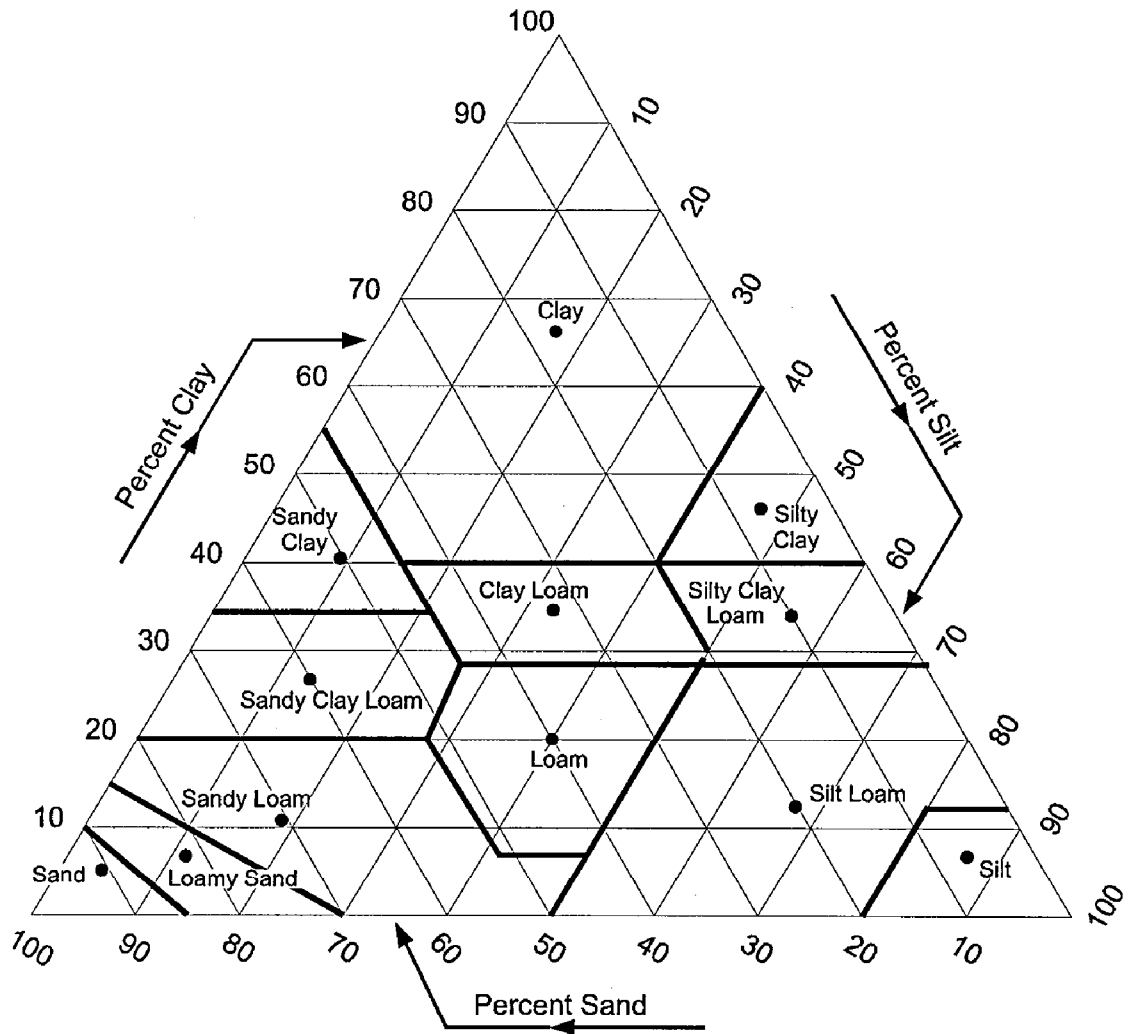


Figure 3. U.S. Soil Conservation Service Classification Chart Showing Centroid Compositions (Solid Circles)

TABLE 3. CENTROID COMPOSITIONS AND MEAN PARTICLE DIAMETERS OF THE 12 SCS SOIL TEXTURAL CLASSIFICATIONS

Textural class	% clay	% silt	% sand	Arithmetic mean particle diameter, cm
Sand	3.33	5.00	91.67	0.044
Loamy sand	6.25	11.25	82.50	0.040
Sandy loam	10.81	27.22	61.97	0.030
Sandy clay loam	26.73	12.56	60.71	0.029
Sandy clay	41.67	6.67	51.66	0.025
Loam	18.83	41.01	40.16	0.020
Clay loam	33.50	34.00	32.50	0.016
Silt loam	12.57	65.69	21.74	0.011
Clay	64.83	16.55	18.62	0.0092
Silty clay loam	33.50	56.50	10.00	0.0056
Silt	6.00	87.00	7.00	0.0046
Silty clay	46.67	46.67	6.66	0.0039

The overall effective diffusion coefficient for systems composed of n distinct soil layers between the source of contamination and the enclosed space floor is:

$$D_T^{\text{eff}} = \frac{L_T}{\sum_{i=0}^n L_i / D_i^{\text{eff}}} \quad (12)$$

where D_T^{eff} = Total overall effective diffusion coefficient, cm^2/s
 L_i = Thickness of soil layer i, cm
 D_i^{eff} = Effective diffusion coefficient across soil layer i, cm^2/s

L_T = Distance between the source of contamination and the bottom of the enclosed space floor, cm.

Note that in the case of cracks in the floor of the enclosed space, the value of L_T does not include the thickness of the floor, nor does the denominator of Equation 12 include the thickness of the floor and the associated effective diffusion coefficient across the crack(s). An unlimited number of soil layers, including the capillary zone, may be included in Equation 12, but all layers must be located between the source of contamination and the enclosed space floor.

2.5 THE INFINITE SOURCE SOLUTION TO CONVECTIVE AND DIFFUSIVE TRANSPORT

Under the assumption that mass transfer is steady-state, Johnson and Ettinger (1991) give the solution for the attenuation coefficient (α) as:

$$\alpha = \frac{\left[\left(\frac{D_T^{eff} A_B}{Q_{building} L_T} \right) \times \exp \left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}} \right) \right]}{\left[\exp \left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}} \right) + \left(\frac{D_T^{eff} A_B}{Q_{building} L_T} \right) + \left(\frac{D_T^{eff} A_B}{Q_{soil} L_T} \right) \left[\exp \left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}} \right) - 1 \right] \right]} \quad (13)$$

where	α	= Steady-state attenuation coefficient, unitless
	D_T^{eff}	= Total overall effective diffusion coefficient, cm ² /s
	A_B	= Area of the enclosed space below grade, cm ²
	$Q_{building}$	= Building ventilation rate, cm ³ /s
	L_T	= Source-building separation, cm
	Q_{soil}	= Volumetric flow rate of soil gas into the enclosed space, cm ³ /s
	L_{crack}	= Enclosed space foundation or slab thickness, cm
	A_{crack}	= Area of total cracks, cm ²

D^{crack} = Effective diffusion coefficient through the cracks, cm^2/s (assumed equivalent to D_i^{eff} of soil layer i in contact with the floor).

The total overall effective diffusion coefficient is calculated by Equation 12. The value of A_B includes the area of the floor in contact with the underlying soil and the total wall area below grade. The building ventilation rate ($Q_{building}$) may be calculated as:

$$Q_{building} = (L_B W_B H_B ER) / 3,600 \text{ s/h} \quad (14)$$

where $Q_{building}$ = Building ventilation rate, cm^3/s
 L_B = Length of building, cm
 W_B = Width of building, cm
 H_B = Height of building, cm
 ER = Air exchange rate, (1/h).

The building dimensions in Equation 14 are those dimensions representing the total "living" space of the building; this assumes that the total air volume within the structure is well mixed and that any vapor contaminant entering the structure is instantaneously and homogeneously distributed.

The volumetric flow rate of soil gas entering the building (Q_{soil}) is calculated by the analytical solution of Nazaroff (1988) such that:

$$Q_{soil} = \frac{2 \pi \Delta P k_v X_{crack}}{\mu \ln(2 Z_{crack} / r_{crack})} \quad (15)$$

where Q_{soil} = Volumetric flow rate of soil gas entering the building, cm^3/s
 π = 3.14159
 ΔP = Pressure differential between the soil surface and the enclosed space, $g/cm-s^2$
 k_v = Soil vapor permeability, cm^2

X_{crack} = Floor-wall seam perimeter, cm

μ = Viscosity of air, g/cm-s

Z_{crack} = Crack depth below grade, cm

r_{crack} = Equivalent crack radius, cm.

Equation 15 is an analytical solution to vapor transport solely by pressure-driven air flow to an idealized cylinder buried some distance (Z_{crack}) below grade; the length of the cylinder is taken to be equal to the building floor-wall seam perimeter (X_{crack}). The cylinder, therefore, represents that portion of the building below grade through which vapors pass. The equivalent radius of the floor-wall seam crack (r_{crack}) is given in Johnson and Ettinger (1991) as:

$$r_{crack} = \eta (A_B / X_{crack}) \quad (16)$$

where r_{crack} = Equivalent crack radius, cm

η = A_{crack}/A_B , ($0 \leq \eta \leq 1$)

A_B = Area of the enclosed space below grade, cm²

X_{crack} = Floor-wall seam perimeter, cm.

The variable r_{crack} is actually the product of the fixed crack-to-total area ratio (η) and the hydraulic radius of the idealized cylinder, which is equal to the total area (A_B) divided by that portion of the cylinder perimeter in contact with the soil gas (X_{crack}). Therefore, if the dimensions of the enclosed space below grade (A_B) and/or the floor-wall seam perimeter (X_{crack}) vary, and the crack-to-total area ratio (η) remains constant, the value of r_{crack} must also vary. The total area of cracks (A_{crack}) is the product of η and A_B .

Equation 15 requires that the soil column properties within the zone of influence of the building (e.g., porosities, bulk density, etc.) be homogeneous, that the soil be isotropic with respect to vapor permeability, and that the pressure within the building be less than atmospheric.

Equation 13 contains the exponent of following dimensionless group:

$$\left(\frac{Q_{soil} L_{crack}}{D_{crack} A_{crack}} \right) . \quad (17)$$

This dimensionless group represents the equivalent Peclet number for transport through the building foundation. As the value of this group approaches infinity, the value of α approaches:

$$\frac{\left(\frac{D_T^{eff} A_B}{Q_{building} L_T} \right)}{\left(\frac{D_T^{eff} A_B}{Q_{soil} L_T} \right) + 1} . \quad (18)$$

In the accompanying spreadsheets, if the exponent of Equation 17 is too great to be calculated, the value of α is set equal to Equation 18.

With a calculated value of α , the steady-state vapor-phase concentration of the contaminant in the building ($C_{building}$) is calculated as:

$$C_{building} = \alpha C_{source} . \quad (19)$$

2.6 THE FINITE SOURCE SOLUTION TO CONVECTIVE AND DIFFUSIVE TRANSPORT

If the thickness of soil contamination is known, the finite source solution of Johnson and Ettinger (1991) can be employed such that the time-averaged attenuation coefficient ($\langle\alpha\rangle$) may be calculated as:

$$\langle\alpha\rangle = \frac{\rho_b C_R \Delta H_c A_B}{Q_{building} C_{source} \tau} \left(\frac{L_T^0}{\Delta H_c} \right) \left[(\beta^2 + 2 \Psi \tau)^{1/2} - \beta \right] \quad (20)$$

where	$\langle\alpha\rangle$	= Time-averaged finite source attenuation coefficient, unitless
	ρ_b	= Soil dry bulk density at the source of contamination, g/cm ³
	C_R	= Initial soil concentration, g/g
	ΔH_c	= Initial thickness of contamination, cm
	A_B	= Area of enclosed space below grade, cm ²
	$Q_{building}$	= Building ventilation rate, cm ³ /s
	C_{source}	= Vapor concentration at the source of contamination, g/cm ³ -v
	τ	= Exposure interval, s
	L_T^0	= Source-building separation at time = 0, cm

and,

$$\beta = \left(\frac{D_T^{eff} A_B}{L_T^0 Q_{soil}} \right) \left[1 - \exp \left(- \frac{Q_{soil} L_{crack}}{D_{crack} A_{crack}} \right) \right] + 1 \quad (21)$$

and,

$$\Psi = \frac{D_T^{eff} C_{source}}{(L_T^0)^2 \rho_b C_R} \quad (22)$$

Implicit in Equation 20 is the assumption that source depletion occurs from the top boundary of the contaminated zone as contaminant volatilizes and moves upward toward the soil surface. This creates a hypothetical "dry zone" (δ) which grows with time; conversely, the "wet zone" of contamination retreats proportionally. When the thickness of the depletion zone (δ) is equal to the initial thickness of contamination (ΔH_c), the source is totally depleted. The unitless expression $(L_T^0/\Delta H_c)[(\beta^2 + 2\Psi\tau)^{1/2} - \beta]$ in Equation 20 represents the cumulative fraction of the depletion zone at the end of the exposure interval τ . Multiplying this expression by the remainder of Equation 20 results in the time-averaged finite source attenuation coefficient ($\langle\alpha\rangle$).

With a calculated value for $\langle\alpha\rangle$, the time-averaged vapor concentration in the building (C_{building}) is:

$$C_{\text{building}} = \langle\alpha\rangle C_{\text{source}} \quad (23)$$

For extended exposure intervals (e.g., 30 years), the time for source depletion may be less than the exposure interval. The time for source depletion (τ_D) may be calculated by:

$$\tau_D = \frac{[\Delta H_c / L_T^0 + \beta]^2 - \beta^2}{2\Psi} \quad (24)$$

If the exposure interval (τ) is greater than the time for source depletion (τ_D), the time-averaged building vapor concentration may be calculated by a mass balance such that:

$$C_{building} = \frac{\rho_b C_R \Delta H_c A_B}{Q_{building} \tau} \quad (25)$$

where	$C_{building}$	= Time-averaged vapor concentration in the building, g/cm ³ -v
	ρ_b	= Soil dry bulk density at the source of contamination, g/cm ³
	C_R	= Initial soil concentration, g/g
	ΔH_c	= Initial thickness of contamination, cm
	A_B	= Area of enclosed space below grade, cm ²
	$Q_{building}$	= Building ventilation rate, cm ³ /s
	τ	= Exposure interval, s.

2.7 THE SOIL GAS MODELS

Use of the Johnson and Ettinger model has typically relied on a theoretical partitioning of the total volume soil concentration into the sorbed, aqueous, and vapor phases. The model has also relied on a theoretical approximation of vapor transport by diffusion and convection from the source of emissions to the building floor in contact with the soil. Use of measured soil gas concentrations directly beneath the building floor instead of theoretical vapor concentrations and vapor transport has obvious advantages that would help to reduce the uncertainty in the indoor air concentration estimates made by the model.

The new soil gas models (SG-SCREEN and SG-ADV) are designed to allow the user to input measured soil gas concentration and sampling depth information directly into the spreadsheets. In the new models, the value of the user-defined soil gas concentration is assigned as the value of C_{source} in Equation 19. The steady-state (infinite source) attenuation coefficient (α) in Equation 19 is calculated using Equation 13. The steady-state solution for the attenuation coefficient is used because no evaluation has been made

regarding the size and total mass of the source of emissions. The source of emissions, therefore, cannot be depleted over time. The new soil gas models estimate the steady-state indoor air concentration over the exposure duration. For a detailed discussion of using the soil gas models as well as soil gas sampling, see Section 4 of this document.

2.8 SOIL VAPOR PERMEABILITY

Soil vapor permeability (k_v) is one of the most sensitive model parameters associated with convective transport of vapors within the zone of influence of the building. Soil vapor permeability is typically measured from field pneumatic tests. If field data are lacking, however, an estimate of the value of k_v can be made with limited data.

Soil intrinsic permeability is a property of the medium alone that varies with the size and shape of connected soil pore openings. Intrinsic permeability (k_i) can be estimated from the soil saturated hydraulic conductivity:

$$k_i = \frac{K_s \mu_w}{\rho_w g} \quad (26)$$

where

- k_i = Soil intrinsic permeability, cm^2
- K_s = Soil saturated hydraulic conductivity, cm/s
- μ_w = Dynamic viscosity of water, g/cm-s ($= 0.01307$ at 10°C)
- ρ_w = Density of water, g/cm^3 ($= 0.999$)
- g = Acceleration due to gravity, cm/s^2 ($= 980.665$).

Schaap and Leij (1998) computed the SCS class average values of the saturated hydraulic conductivity (K_s) for each of the 12 SCS soil textural classifications (Table 4). These data replace the mean values developed by Carsel and Parrish (1988) included in the previous EPA versions of the Johnson and Ettinger models. With these values, a general estimate of the value of k_i can be made by soil type. As an alternative, in situ measurements of the site-specific saturated hydraulic conductivity can be made and the results input into Equation 26 to compute the value of the soil intrinsic permeability.

TABLE 4. CLASS AVERAGE VALUES OF SATURATED HYDRAULIC CONDUCTIVITY FOR THE 12 SCS SOIL TEXTURAL CLASSIFICATIONS

Soil texture , USDA	Class average saturated hydraulic conductivity, cm/h
Sand	26.78
Loamy sand	4.38
Sandy loam	1.60
Sandy clay loam	0.55
Sandy clay	0.47
Loam	0.50
Clay loam	0.34
Silt loam	0.76
Clay	0.61
Silty clay loam	0.46
Silt	1.82
Silty clay	0.40

Effective permeability is the permeability of the porous medium to a fluid when more than one fluid is present; it is a function of the degree of saturation. The relative air permeability of soil (k_{rg}) is the effective air permeability divided by the intrinsic permeability and therefore takes into account the effects of the degree of water saturation on air permeability.

Parker et al. (1987) extended the relative air permeability model of van Genuchten (1980) to allow estimation of the relative permeabilities of air and water in a two- or three-phase system:

$$k_{rg} = (1 - S_{le})^{1/2} (1 - S_{le}^{1/M})^{2M} \quad (27)$$

where k_{rg} = Relative air permeability, unitless ($0 \leq k_{rg} \leq 1$)

S_{te} = Effective total fluid saturation, unitless

M = van Genuchten shape parameter, unitless.

Given a two-phase system (i.e., air and water), the effective total fluid saturation (S_{te}) is calculated as:

$$S_{te} = \frac{(\theta_w - \theta_r)}{(n - \theta_r)} \quad (28)$$

where S_{te} = Effective total fluid saturation, unitless

θ_w = Soil water-filled porosity, cm^3/cm^3

θ_r = Residual soil water content, cm^3/cm^3

n = Soil total porosity, cm^3/cm^3 .

Class average values for the parameters θ_r and M by SCS soil type may be obtained from Table 2.

The effective air permeability (k_v) is then the product of the intrinsic permeability (k_i) and the relative air permeability (k_{rg}) at the soil water-filled porosity θ_w .

2.8 CALCULATION OF A RISK-BASED SOIL OR GROUNDWATER CONCENTRATION

Both the infinite source model estimate of the steady-state building concentration and the finite source model estimate of the time-averaged building concentration represent the exposure point concentration used to assess potential risks.

Calculation of a risk-based media concentration for a carcinogenic contaminant takes the form:

$$C_c = \frac{TR \times AT_c \times 365 \text{ days/yr}}{URF \times EF \times ED \times C_{\text{building}}} \quad (29)$$

where	C_C	= Risk-based media concentration for carcinogens, $\mu\text{g/kg-soil}$, or $\mu\text{g/L-water}$
	TR	= Target risk level, unitless
	AT_C	= Averaging time for carcinogens, yr
	URF	= Unit risk factor, $(\mu\text{g/m}^3)^{-1}$
	EF	= Exposure frequency, days/yr
	ED	= Exposure duration, yr
	C_{building}	= Vapor concentration in the building, $\mu\text{g/m}^3$ per $\mu\text{g/kg-soil}$, or $\mu\text{g/m}^3$ per $\mu\text{g/L-water}$.

In the case of a noncarcinogenic contaminant, the risk-based media concentration is calculated by:

$$C_{NC} = \frac{THQ \times AT_{NC} \times 365 \text{ days / yr}}{EF \times ED \times \frac{1}{RfC} \times C_{\text{building}}} \quad (30)$$

where	C_{NC}	= Risk-based media concentration for noncarcinogens, $\mu\text{g/kg-soil}$, or $\mu\text{g/L-water}$
	THQ	= Target hazard quotient, unitless
	AT_{NC}	= Averaging time for noncarcinogens, yr
	EF	= Exposure frequency, days/yr
	ED	= Exposure duration, yr
	RfC	= Reference concentration, mg/m^3
	C_{building}	= Vapor concentration in the building, mg/m^3 per $\mu\text{g/kg-soil}$, or mg/m^3 per $\mu\text{g/L-water}$.

The accompanying spreadsheets calculate risk-based media concentrations based on a unity initial concentration. That is, soil risk-based concentrations are calculated with

an initial hypothetical soil concentration of 1 µg/kg-soil, while for groundwater the initial hypothetical concentration is 1 µg/L-water.

For this reason, the values of C_{source} and $C_{building}$ shown on the INTERCALCS worksheet when reverse-calculating a risk-based media concentration do not represent actual values. For these calculations, the following message will appear on the RESULTS worksheet:

"MESSAGE: The values of C_{source} and $C_{building}$ on the INTERCALCS worksheet are based on unity and do not represent actual values."

When forward-calculating risks from a user-defined initial soil or groundwater concentration, the values of C_{source} and $C_{building}$ on the INTERCALCS worksheet are correct.

2.10 CALCULATION OF INCREMENTAL RISKS

Forward-calculation of incremental risks begins with an actual initial media concentration (i.e., µg/kg-soil or µg/L-water). For carcinogenic contaminants, the risk level is calculated as:

$$Risk = \frac{URF \times EF \times ED \times C_{building}}{AT_C \times 365 \text{ days / yr}} \quad (31)$$

For noncarcinogenic contaminants, the hazard quotient (HQ) is calculated as:

$$HQ = \frac{EF \times ED \times \frac{1}{RfC} \times C_{building}}{AT_{NC} \times 365 \text{ days / yr}} \quad (32)$$

2.11 MAJOR MODEL ASSUMPTIONS/LIMITATIONS

The following represent the major assumptions/limitations of the Johnson and Ettinger model. Additional assumptions specific to the application of the model as applied in the accompanying spreadsheets are contained in Section 5.

1. Contaminant vapors enter the structure primarily through cracks and openings in the walls and foundation.
2. Convective transport occurs primarily within the building zone of influence and vapor velocities decrease rapidly with increasing distance from the structure.
3. Diffusion dominates vapor transport between the source of contamination and the building zone of influence.
4. All vapors originating from below the building will enter the building unless the floors and walls are perfect vapor barriers.
5. All soil properties in any horizontal plane are homogeneous.
6. The contaminant is homogeneously distributed within the zone of contamination.
7. The areal extent of contamination is greater than that of the building floor in contact with the soil.
8. Vapor transport occurs in the absence of convective water movement within the soil column (i.e., evaporation or infiltration), and in the absence of mechanical dispersion.
9. The model does not account for transformation processes (e.g., biodegradation, hydrolysis, etc.).
10. The soil layer in contact with the structure floor and walls is isotropic with respect to permeability.
11. Both the building ventilation rate and the difference in dynamic pressure between the interior of the structure and the soil surface are constant values.

SECTION 3

SOIL AND GROUNDWATER MODEL APPLICATION

This section provides step-by-step instructions on how to implement the soil and groundwater contamination versions of the Johnson and Ettinger model using the accompanying spreadsheets. Section 4 discusses application of the soil gas versions of the model. The user provides data and selects certain input options, and views model results via a series of worksheets. Error messages are provided within both the data entry worksheet and the results worksheet to warn the user that entered data are missing or outside of permitted limits.

3.1 RUNNING THE MODELS

Four different models are provided as both MICROSOFT EXCEL and LOTUS 1-2-3 spreadsheets.

1. Screening-Level Models for Soil Contamination:
 - SL-SCREEN.XLS (EXCEL)
 - SL-SCREEN.WK4 (1-2-3)
2. Screening-Level Models for Groundwater Contamination:
 - GW-SCREEN.XLS (EXCEL)
 - GW-SCREEN.WK4 (1-2-3)
3. Advanced Models for Soil Contamination:
 - SL-ADV.XLS (EXCEL)
 - SL-ADV.WK4 (1-2-3)

4. Advanced Models for Groundwater Contamination:

- GW-ADV.XLS (EXCEL)
- GW-ADV.WK4 (1-2-3).

Both the screening-level models and the advanced models allow the user to calculate a risk-based media concentration or incremental risks from an actual starting concentration in soil or in groundwater. Data entry within the screening-level models is limited to the most sensitive model parameters and incorporates only one soil stratum above the contamination. The advanced models provide the user with the ability to enter data for all of the model parameters and also incorporates up to three individual soil strata above the contamination for which soil properties may be varied.

To run any of the models, simply open the appropriate model file within either MICROSOFT EXCEL or LOTUS 1-2-3. Each model is constructed of the following worksheets:

1. DATENTER (Data Entry Sheet)
2. CHEMPROPS (Chemical Properties Sheet)
3. INTERCALCS (Intermediate Calculations Sheet)
4. RESULTS (Results Sheet)
5. VLOOKUP (Lookup Tables)

The following is an explanation of what is contained in each worksheet, how to enter data, how to interpret model results, and how to add/revise the chemical properties data found in the VLOOKUP Tables. As examples, Appendix A contains all the worksheets for the advanced soil contamination model SL-ADV.

Note: Because of the limitations of LOTUS 1-2-3, variable names (e.g., ΔH_c) appear in alphanumeric characters. Subscripts are preceded by the symbol "~" and superscripts are preceded by the symbol "^." Upper case Greek characters are spelled with an initial capital letter and lower case Greek characters are spelled in lower case. For example, the variable ΔH_c would appear as "Delta H~c" while the variable D_T^{eff} would appear as "D~T^eff."

3.2 THE DATA ENTRY SHEET (DATENTER)

Figure 4 is an example of a data entry sheet. In this case, it shows the data entry sheet for the screening-level model for contaminated groundwater (GW-SCREEN). Figure 5 is an example of an advanced model data entry sheet (GW-ADV). Note that the screening-level model sheet requires entry of considerably less data than does the advanced sheet. To enter data, simply position the cursor within the appropriate box and type the value; all other cells are protected.

3.2.1 Error Messages

In the case of the screening-level models, all error messages will appear in red type below the applicable row of data entry boxes. For the advanced models, error messages may appear on the data entry sheet or in the lower portion of the results sheet. Error messages will occur if required entry data are missing or if data are out of range or do not conform to model conventions. The error message will tell the user what kind of error has occurred.

Figure 6 is an example of an error message appearing on the data entry sheet. Figure 7 illustrates error messages appearing within the message and error summary section on the results sheet (advanced models only).

3.2.2 Entering Data

Each data entry sheet requires the user to input values for model variables. Data required for the soil contamination scenario will differ from that required for the groundwater contamination scenario. In addition, data required for the screening-level models will differ from that required for the advanced models.

CALCULATE RISK-BASED GROUNDWATER CONCENTRATION (enter "X" in "YES" box)

YES ☒

OR

CALCULATE INCREMENTAL RISKS FROM ACTUAL GROUNDWATER CONCENTRATION
(enter "X" in "YES" box and initial groundwater conc. below)

YES ☐

ENTER Chemical CAS No. (numbers only, no dashes)	ENTER Initial groundwater conc., C _w (µg/L)	Chemical	
56235		Carbon tetrachloride	
ENTER Depth below grade to bottom of enclosed space floor, L _F (cm)	ENTER Depth below grade to water table, L _{WT} (cm)	ENTER SCS soil type directly above water table	ENTER Average soil/ groundwater temperature, T _s (°C)
200	400	SC	10

ENTER Vadose zone SCS soil type (used to estimate soil vapor permeability)	OR	ENTER User-defined vadose zone soil vapor permeability, k _v (cm ²)	ENTER Vadose zone soil dry bulk density, ρ _b ^v (g/cm ³)	ENTER Vadose zone soil total porosity, n _v (unitless)	ENTER Vadose zone soil water-filled porosity, θ _w ^v (cm ³ /cm ³)
SC			1.5	0.43	0.3

ENTER Target risk for carcinogens, TR (unitless)	ENTER Target hazard quotient for noncarcinogens, THQ (unitless)	ENTER Averaging time for carcinogens, AT _c (yrs)	ENTER Averaging time for noncarcinogens, AT _{NC} (yrs)	ENTER Exposure duration, ED (yrs)	ENTER Exposure frequency, EF (days/yr)
1.0E-06	1	70	30	30	350
Used to calculate risk-based groundwater concentration.					

Figure 4. GW-SCREEN Data Entry Sheet

CALCULATE RISK-BASED GROUNDWATER CONCENTRATION (enter "X" in "YES" box)

YES ☒

OR

CALCULATE INCREMENTAL RISKS FROM ACTUAL GROUNDWATER CONCENTRATION (enter "X" in "YES" box and initial groundwater conc. below)

YES ☐

ENTER Chemical CAS No. (numbers only, no dashes)		ENTER Initial groundwater conc., C_w (mg/L)		Chemical Carbon tetrachloride							
56235											
ENTER Average soil groundwater temperature, T_a (°C)	ENTER Depth below grade to bottom of enclosed space floor, L_a (cm)	ENTER Depth below grade to water table, L_{wt} (cm)	ENTER Totals must add up to value of L_{wt} (cell D28) Thickness of soil stratum A, h_a (cm)			ENTER Thickness of soil stratum B, h_b (cm) (Enter value or 0)	ENTER Thickness of soil stratum C, h_c (cm) (Enter value or 0)	ENTER Soil stratum directly above water table, (Enter A, B, or C)	ENTER SCS soil type directly above water table	ENTER Soil stratum A SCS soil type (used to estimate soil vapor permeability)	ENTER User-defined stratum A soil vapor permeability, k_a (cm²)
10	200	400	300	50	50			C	SC	C	
ENTER Stratum A soil dry bulk density, ρ_b^A (g/cm³)	ENTER Stratum A soil total porosity, n^A (unitless)	ENTER Stratum A soil water-filled porosity, θ_w^A (cm³/cm³)	ENTER Stratum B soil dry bulk density, ρ_b^B (g/cm³)	ENTER Stratum B soil total porosity, n^B (unitless)	ENTER Stratum B soil water-filled porosity, θ_w^B (cm³/cm³)	ENTER Stratum C soil dry bulk density, ρ_b^C (g/cm³)	ENTER Stratum C soil total porosity, n^C (unitless)	ENTER Stratum C soil water-filled porosity, θ_w^C (cm³/cm³)			
1.5	0.43	0.2	1.7	0.42	0.27	1.7	0.43	0.3			
ENTER Enclosed space floor thickness, L_{boca} (cm)	ENTER Soil-bldg. pressure differential, ΔP (g/cm-s²)	ENTER Enclosed space floor length, L_b (cm)	ENTER Enclosed space floor width, W_b (cm)	ENTER Enclosed space height, H_b (cm)	ENTER Floor-wall seam crack width, w (cm)	ENTER Indoor air exchange rate, ER (1/h)					
15	40	961	961	488	0.1	0.45					
ENTER Averaging time for carcinogens, AT_c (yrs)	ENTER Averaging time for noncarcinogens, AT_{nc} (yrs)	ENTER Exposure duration, ED (yrs)	ENTER Exposure frequency, EF (days/yr)	ENTER Target risk for carcinogens, TR (unitless)	ENTER Target hazard quotient for noncarcinogens, THQ (unitless)						
70	30	30	350	1.0E-06	1						
Used to calculate risk-based groundwater concentration.											

Figure 5. GW-ADV Data Entry Sheet

CALCULATE RISK-BASED GROUNDWATER CONCENTRATION (enter "X" in "YES" box)

YES ☒

OR

CALCULATE INCREMENTAL RISKS FROM ACTUAL GROUNDWATER CONCENTRATION
(enter "X" in "YES" box and initial groundwater conc. below)

YES ☒

ENTER	ENTER	
Chemical	Initial	Cannot calculate risk-based concentration and incremental risk simultaneously.
CAS No.	groundwater	
(numbers only,	conc.,	
no dashes)	C _w	
	(µg/L)	Chemical

Figure 6. Example Error Message on Data Entry Sheet

RISK-BASED SOIL CONCENTRATION CALCULATIONS:

INCREMENTAL RISK CALCULATIONS:

Indoor exposure soil conc., carcinogen (µg/kg)	Indoor exposure soil conc., noncarcinogen (µg/kg)	Risk-based indoor exposure soil conc., (µg/kg)	Soil saturation conc., C _{sat} (µg/kg)	Final indoor exposure soil conc., (µg/kg)	Incremental risk from vapor intrusion to indoor air, carcinogen (unitless)	Hazard quotient from vapor intrusion to indoor air, noncarcinogen (unitless)
NA	NA	NA	4.81E+05	NA	2.9E-05	NA

ERROR SUMMARY BELOW: (DO NOT USE RESULTS IF ERRORS ARE PRESENT)

Combined thickness of strata A + B + C must be = depth below grade to top of contamination.

Figure 7. Example Error Message on Results Sheet

Model Variables--

The following is a list of all data entry variables required for evaluating either a risk-based media concentration or the incremental risks due to actual contamination. A description for which model(s) the variable is appropriate is given in parenthesis after the name of the variable. In addition, notes on how the variable is used in the calculations, and how to determine appropriate values of the variable are given below the variable name. A quick determination of which variables are required for a specific model can be made by reviewing the data entry sheet for the model chosen. Example data entry sheets for each model can be found in Appendix B.

1. *Calculate Risk-Based Concentration OR Calculate Incremental Risks from Actual Concentration (All Soil and Groundwater Models)*

The model will calculate either a risk-based soil or groundwater concentration or incremental risks but cannot calculate both simultaneously. Enter an "X" in only one box.

2. *Chemical CAS No. (All Models)*

Enter the appropriate CAS number for the chemical you wish to evaluate; do not enter dashes. The CAS number entered must exactly match that of the chemical or the error message "CAS No. not found" will appear in the "Chemical" box. Once the correct CAS number is entered, the name of the chemical will automatically appear in the "Chemical" box. A total of 93 chemicals and their associated properties are included with each model; see Section 3.7 for instructions on adding/revising chemicals.

3. *Initial Soil or Groundwater Concentration (All Soil and Groundwater Models)*

Enter a value only if incremental risks are to be calculated. Be sure to enter the concentration in units of $\mu\text{g/kg}$ (soil) or $\mu\text{g/L}$ (groundwater). Typically, this value represents the average concentration within the zone of contamination. If descriptive statistics are not available to quantify the uncertainty in the average value, the maximum value may be used as an upper bound estimate.

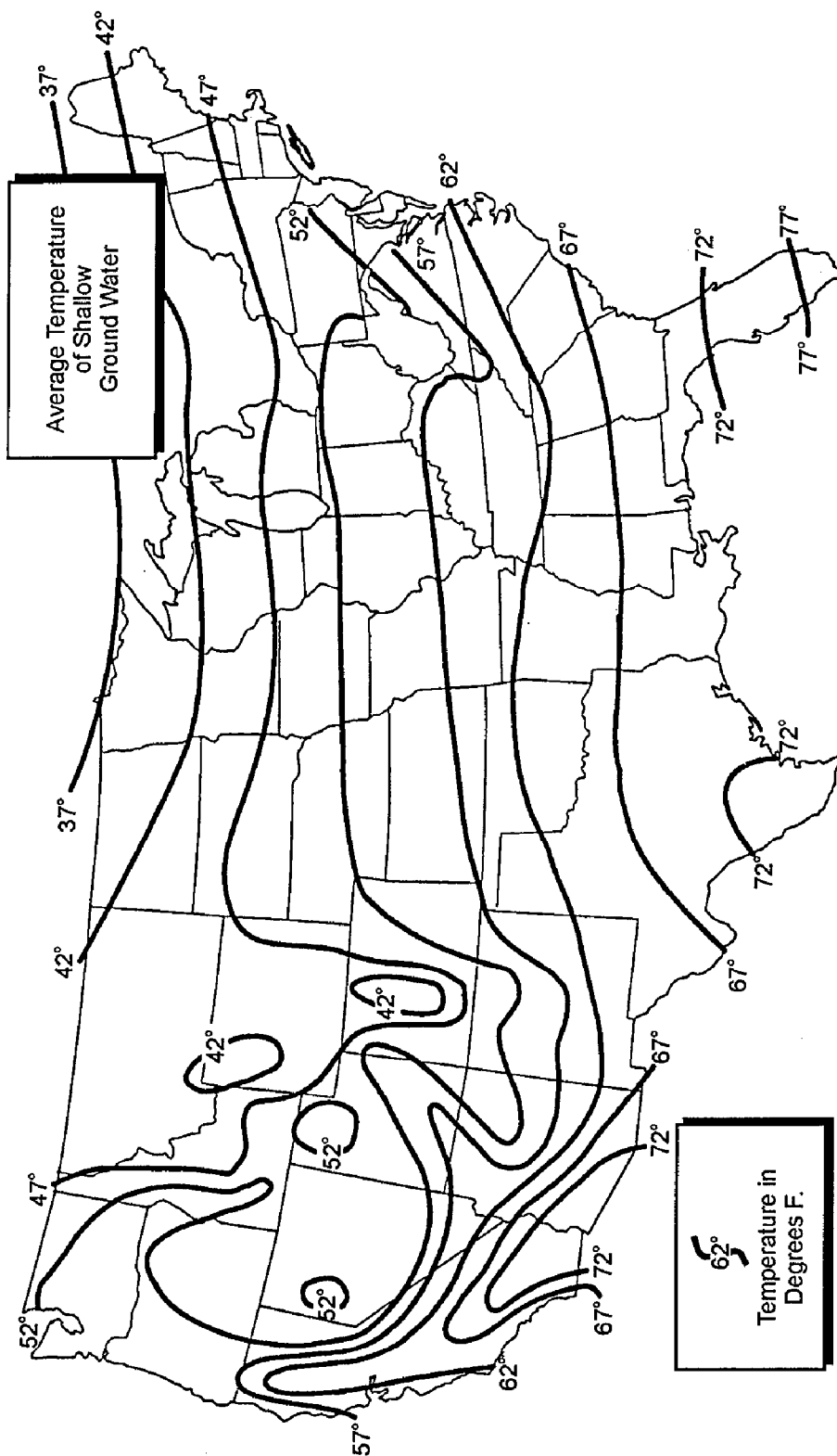


Figure 8. Average Shallow Groundwater Temperature in the United States

4. *Average Soil/Groundwater Temperature (All Models)*

The soil/groundwater temperature is used to correct the Henry's law constant to the specified temperature. Figure 8 from U.S. EPA (1995) shows the average temperature of shallow groundwater in the continental United States. Shallow groundwater temperatures may be used to approximate subsurface soil temperatures greater than one to two meters below the ground surface. Another source of information may be your State groundwater protection regulatory agency.

5. *Depth Below Grade to Bottom of Enclosed Space Floor (All Models)*

Enter the depth to the bottom of the floor in contact with the soil. The default value for slab-on-grade and basement construction is 15 cm and 200 cm, respectively.

6. *Depth Below Grade to Top of Contamination (Soil Models Only)*

Enter the depth to the top of soil contamination. If the contamination begins at the soil surface, enter the depth below grade to the bottom of the enclosed space floor. The depth to the top of contamination must be greater than or equal to the depth to the bottom of the floor.

7. *Depth Below Grade to Water Table (Groundwater Models Only)*

Enter the depth to the top of the water table (i.e., where the pressure head is equal to zero and the pressure is atmospheric).

Note: The thickness of the capillary zone is calculated based on the SCS soil textural classification above the top of the water table. The depth below grade to the top of the water table minus the thickness of the capillary zone must be greater than the depth below grade to the bottom of the enclosed space floor. This means that the top of the capillary zone is always below the floor.

8. *Depth Below Grade to Bottom of Contamination (Advanced Soil Model Only)*

This value is used to calculate the thickness of soil contamination. A value greater than zero and greater than the depth to the top of contamination will automatically invoke the finite source model. If the thickness of contamination is unknown, two options are available:

1. Entering a value of zero will automatically invoke the infinite source model.

2. Enter the depth to the top of the water table. This will invoke the finite source model under the assumption that contamination extends from the top of contamination previously entered down to the top of the water table.

9. *Thickness of Soil Stratum "X" (Advanced Models Only)*

In the advanced models, the user can define up to three soil strata between the soil surface and the top of contamination or to the soil gas sampling depth, as appropriate. These strata are listed as A, B, and C. Stratum A extends down from the soil surface, Stratum B is below Stratum A, and Stratum C is the deepest stratum. The thickness of Stratum A must be at least as thick as the depth below grade to the bottom of the enclosed space floor. The combined thickness of all strata must be equal to the depth to the top of contamination, or to the soil gas sampling depth, as appropriate. If soil strata B and/or C are not to be considered, a value of zero must be entered for each stratum not included in the analysis.

10. *Soil Stratum A SCS Soil Type (Advanced Models Only)*
Enter one of the following SCS soil type abbreviations:

<u>Abbreviation</u>	<u>SCS Soil Type</u>
C	Clay
CL	Clay loam
L	Loam
LS	Loamy sand
S	Sand
SC	Sandy clay
SCL	Sandy clay loam
SI	Silt
SIC	Silty clay
SICL	Silty clay loam
SIL	Silt loam
SL	Sandy loam

The SCS soil textural classification can be determined by using either the American Society for Testing and Materials (ASTM) Standard Test Method for Particle-Size Analysis of Soils (D422-63) or by using the analytical procedures found in the U.S. Natural Resources Conservation Service

(NRCS) Soil Survey Laboratory Methods Manual, Soil Survey Laboratory Investigations Report No. 42. After determining the particle size distribution of a soil sample, the SCS soil textural classification can be determined using the SCS classification chart in Figure 3.

The SCS soil type along with the Stratum A soil water-filled porosity is used to estimate the soil vapor permeability of Stratum A which is in contact with the floor and walls of the enclosed space below grade. Alternatively, the user may define a soil vapor permeability (see Variable No. 11).

11. *User-Defined Stratum A Soil Vapor Permeability (Advanced Models Only)*

As an alternative to estimating the soil vapor permeability of soil Stratum A, the user may define the soil vapor permeability. As a general guide, the following represent the practical range of vapor permeabilities:

<u>Soil type</u>	<u>Soil vapor permeability, cm²</u>
Medium sand	1.0×10^{-7} to 1.0×10^{-6}
Fine sand	1.0×10^{-8} to 1.0×10^{-7}
Silty sand	1.0×10^{-9} to 1.0×10^{-8}
Clayey silts	1.0×10^{-10} to 1.0×10^{-9}

See Section 5.2 for a more detailed discussion.

12. *Vadose Zone SCS Soil Type (Screening Models Only)*

Because the screening-level models accommodate only one soil stratum above the top of contamination or soil gas sampling depth, enter the SCS soil type from the list given in Variable No. 10.

13. *User-Defined Vadose Zone Soil Vapor Permeability (Screening Models Only)*

For the same reason cited in No. 12 above, the user may alternatively define a soil vapor permeability. Use the list of values given in Variable No. 11 as a general guide. See Section 5.2 for a more detailed discussion.

14. *Soil Stratum Directly Above the Water Table (Advanced Groundwater Models Only)*

Enter either A, B, or C as the soil stratum directly above the water table. This value must be the letter of the deepest stratum for which a thickness value has been specified under Variable No. 9.

15. *SCS Soil Type Directly Above Water Table (Groundwater Models Only)*

Enter the correct SCS soil type from the list given in Variable No. 10 for the soil type directly above the water table. The soil type entered is used to estimate the rise (thickness) of the capillary zone.

16. *Stratum "X" Soil Dry Bulk Density (Advanced Models Only)*

Enter the average soil dry bulk density for the appropriate soil stratum. Dry bulk density is used in a number of intermediate calculations and is normally determined by field measurements (ASTM D 2937 Method).

17. *Stratum "X" Soil Total Porosity (Advanced Models Only)*

Total soil porosity (n) is determined as:

$$n = 1 - \rho_b / \rho_s$$

where ρ_b is the soil dry bulk density (g/cm^3) and ρ_s is the soil particle density (usually $2.65 \text{ g}/\text{cm}^3$).

18. *Stratum "X" Soil Water-Filled Porosity (Advanced Models Only)*

Enter the average long-term volumetric soil moisture content; this is typically a depth-averaged value for the appropriate soil stratum. A long-term average value is typically not readily available. Do not use values based on episodic measurements unless they are representative of long-term conditions.

One option is to use a model to estimate the long-term average soil water-filled porosities of each soil stratum between the enclosed space floor and the top of contamination. The HYDRUS model version 5.0 (Vogel et al.,

1996) is a public domain code for simulating one-dimensional water flow, solute transport, and heat movement in variably-saturated soils. The water flow simulation module of HYDRUS will generate soil water content as a function of depth and time given actual daily precipitation data. Model input requirements include either the soil hydraulic properties of van Genuchten (1980) or those of Brooks and Corey (1966). The van Genuchten soil hydraulic properties required are the same as those given in Tables 2 and 4 (i.e., θ_s , θ_r , N , α , and K_s). The HYDRUS model is available from the U.S. Department of Agriculture (USDA) - Agricultural Research Service in Riverside, California via their internet website at <http://www.ussl.ars.usda.gov/MODELS/HYDRUS.HTM>. One and two-dimensional commercial versions of HYDRUS (Windows versions) are available at the International Ground Water Modeling Center website at <http://www.mines.edu/research/igwmc/software/>. Schaap and Leij (1998) have recently developed a Windows program entitled ROSETTA for estimating the van Genuchten soil hydraulic properties based on a limited or more extended set of input data. The ROSETTA program can be found at the USDA website: <http://www.ussl.ars.usda.gov/MODELS/rosetta/rosetta.htm>. The van Genuchten hydraulic properties can then be input into HYDRUS to estimate soil moisture content.

19. *Stratum "X" Soil Organic Carbon Fraction (Advanced Soil Models Only)*

Enter the depth-averaged soil organic carbon fraction for the stratum specified. Soil organic carbon is measured by burning-off soil carbon in a controlled-temperature oven. This parameter, along with the chemical's organic carbon partition coefficient (K_{oc}), is used to determine the soil-water partition coefficient (K_d).

20. *Vadose Zone Soil Dry Bulk Density (Screening Models Only)*

Because the screening-level models accommodate only one soil stratum above the top of contamination, enter the depth-averaged soil dry bulk density. The default value is 1.5 g/cm^3 which is consistent with U.S. EPA (1996a and b) for subsurface soils.

21. *Vadose Zone Soil Total Porosity (Screening Models Only)*

Because the screening-level models accommodate only one soil stratum above the top of contamination, enter the depth-averaged soil total porosity. The default value is 0.43 which is consistent with U.S. EPA (1996a and b) for subsurface soils.

22. *Vadose Zone Soil Water-Filled Porosity (Screening Models Only)*

Because the screening-level models accommodate only one soil stratum above the top of contamination, enter the depth-averaged soil water-filled porosity. The default value is 0.30 which is consistent with U.S. EPA (1996a and b) for subsurface soils.

23. *Vadose Zone Soil Organic Carbon Fraction (Soil Screening Model Only)*

Because the screening-level models accommodate only one soil stratum above the top of contamination, enter the depth-averaged soil organic carbon fraction. The default value is 0.002 which is consistent with U.S. EPA (1996a and b) for subsurface soils.

24. *Enclosed Space Floor Thickness (Advanced Models Only)*

Enter the thickness of the floor slab. All models operate under the assumption that the floor in contact with the underlying soil is composed of impermeable concrete whether constructed as a basement floor or slab-on-grade. The default value is 15 cm which is consistent with Johnson and Ettinger (1991).

25. *Soil-Building Pressure Differential (Advanced Models Only)*

Because of wind effects on the structure, stack effects due to heating of the interior air, and unbalanced mechanical ventilation, a negative pressure with respect to the soil surface is generated within the structure. This pressure differential (ΔP) induces a flow of soil gas through the soil matrix and into the structure through cracks, gaps, and openings in the foundation. The effective range of values of ΔP is 0-20 Pascals (Loureiro et al., 1990; Eaton and Scott, 1984). Individual average values for wind effects and stack effects are approximately 2 Pa (Nazaroff et al., 1985; Put and Meijer, 1989). Typical values for the combined effects of wind pressures and heating are 4 to 5 Pa (Loureiro et al., 1990; Grimsrud et al., 1983). A conservative default value of ΔP was therefore chosen to be 4 Pa (40 g/cm-s²).

For more information on estimating site-specific values of ΔP , the user is referred to Nazaroff et al. (1987) and Grimsrud et al. (1983).

26. *Enclosed Space Floor Length (Advanced Models Only)*

The default value is 961 cm (see Variable No. 28).

27. *Enclosed Space Floor Width (Advanced Models Only)*

The default value is 961 cm (see Variable No. 28).

28. *Enclosed Space Height (Advanced Models Only)*

The default values of the enclosed space length, width, and height were derived from the average estimated volume of both owner-occupied and rental single-family detached residences in the U.S. (U.S. DOE, 1995). These dimensions assume a living space volume of 451 m^3 divided into two stories of equal volume with ceiling heights of 8 ft (2.44 m). Each floor is assumed to be a square with a total floor area of 92.42 m^2 and equal lengths and widths of 9.61 m. The total height of the living space is 4.88 m.

29. *Floor-Wall Seam Crack Width (Advanced Models Only)*

The conceptual model used in the spreadsheets follows that of Loureiro et al. (1990) and Nazaroff (1988) and is illustrated in Figure 9. The model is based on a single-family house with a poured concrete basement floor and wall foundations, or constructed slab-on-grade in similar fashion. A gap is assumed to exist at the junction between the floor and the foundation along the perimeter of the floor. The gap exists as a result of building design or concrete shrinkage. This gap is assumed to be the only opening in the understructure of the house and therefore the only route for soil gas entry. Eaton and Scott (1984) reported typical open areas of joints between wall and floor slabs of residential structures in Canada of approximately 300 cm^2 . Therefore, given the default floor length and width of 961 cm, a gap width (w) of 0.1 cm equates to a total gap area of 384 cm^2 , which is reasonable given the findings of Eaton and Scott. This value of the gap width is also consistent with the typical value reported in Loureiro et al. (1990). The default value of the floor-wall seam crack width was therefore set equal to 0.1 cm.

30. *Indoor Air Exchange Rate (Advanced Models Only)*

The indoor air exchange rate is used, along with the building dimensions, to calculate the building ventilation rate. The default value of the indoor air exchange rate is 0.45/h. This value is consistent with both the geometric

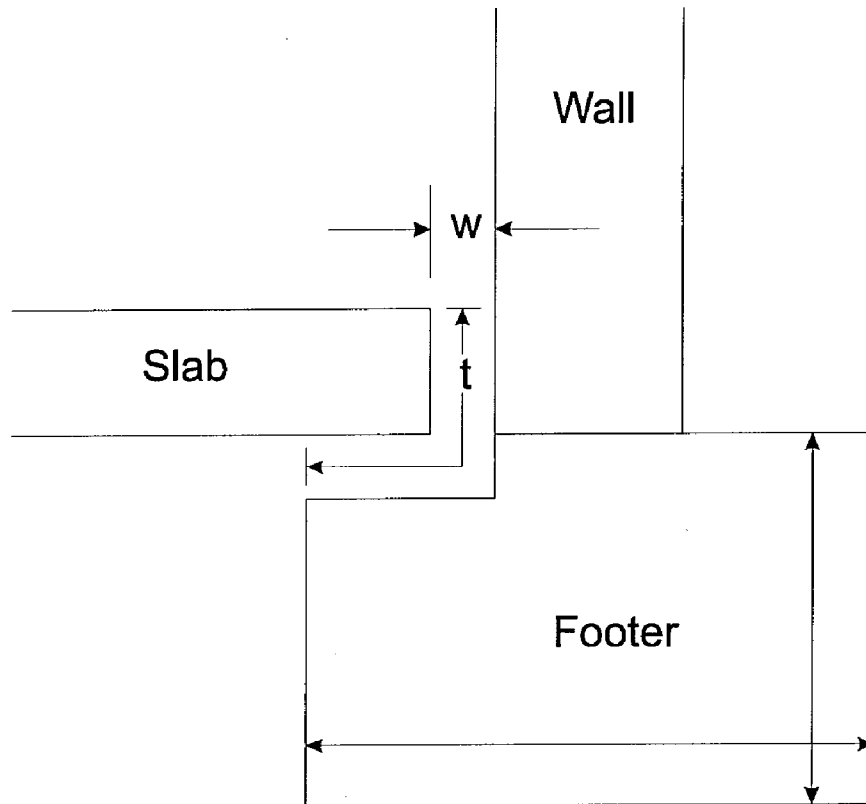


Figure 9. Floor Slab and Foundation

mean and the 50th percentile of houses in all regions of the U.S. as reported in Koontz and Rector (1995). This value is also consistent with the average of the control group of 331 houses in a study conducted by Parker et al. (1990) to compare data with that of 292 houses with energy-efficient features in the Pacific Northwest.

31. *Averaging Time for Carcinogens (All Models)*

Enter the averaging time in units of years. The default value is 70 years.

32. *Averaging Time for Noncarcinogens (All Models)*

Enter the averaging time in units of years. The averaging time for noncarcinogens is set equal to the exposure duration. The default value for residential exposure from U.S. EPA (1996a and b) is 30 years.

33. *Exposure Duration (All Models)*

Enter the exposure duration in units of years. The default value for residential exposure from U.S. EPA (1996a and b) is 30 years.

34. *Exposure Frequency (All Models)*

Enter the exposure frequency in units of days/yr. The default value for residential exposure from U.S. EPA (1996a and b) is 350 days/yr.

35. *Target Risk for Carcinogens (All Soil and Groundwater Models)*

If a risk-based media concentration is to be calculated, enter the target risk-level. The default value is 1×10^{-6} .

36. *Target Hazard quotient for Noncarcinogens (All Soil and Groundwater Models)*

If a risk-based media concentration is to be calculated, enter the target hazard quotient. The default value is 1.

The remaining four worksheets include the results sheet (RESULTS) and three ancillary sheets. The ancillary sheets include the chemical properties sheet (CHEMPROPS), the intermediate calculations sheet (INTERCALCS), and the lookup tables (VLOOKUP).

3.3 THE RESULTS SHEET (RESULTS)

Once all data are entered in the data entry sheet, the model results may be viewed on the RESULTS sheet. For the soil and groundwater models, calculations are presented as either a risk-based soil or groundwater concentration, or the incremental risks associated with an initial soil or groundwater concentration. In the case of the advanced models, the user should check the message and error summary below the results section to ensure that no error messages appear. If one or more error messages appear, re-enter the appropriate data.

The RESULTS worksheet shows the indoor exposure soil or groundwater concentration for either a carcinogen or noncarcinogen as appropriate. When a contaminant is both a carcinogen and a noncarcinogen, the risk-based indoor exposure concentration is set equal to the lower of these two values. In addition, the soil saturation concentration (C_{sat}) or the aqueous solubility limit (S) is also displayed for the soil and groundwater models, respectively.

The equilibrium vapor concentration at the source of contamination is limited by the value of C_{sat} for soil contamination and by the value of S for groundwater contamination, as appropriate. For a single contaminant, the vapor concentration directly above the source of soil contamination cannot be greater than that associated with the soil saturation concentration; for groundwater contamination, the vapor concentration cannot be greater than that associated with the solubility limit. As a result, subsurface soil concentrations greater than C_{sat} and groundwater concentrations greater than S will not produce higher vapor concentrations. Therefore, if the indoor vapor concentration predicted from a soil concentration greater than or equal to the value of C_{sat} does not exceed the health-based limit in indoor air (target risk or target hazard quotient), the vapor intrusion pathway will not be of concern for that particular chemical. The same is true for an indoor vapor concentration predicted from a groundwater concentration greater than or equal to the value of S . That does not necessarily mean, however, that the subsurface contamination

will not be of concern from a groundwater protection standpoint, and the potential for free-phase contamination (e.g., NAPL) must also be addressed.

For subsurface soils, the physical state of a contaminant at the soil temperature plays a significant role. When a contaminant is a liquid (or gas) at the soil temperature, the upper limit of the soil screening level is set at C_{sat} . This tends to reduce the potential for NAPL to exist within the vadose zone. The case is different for a subsurface contaminant that is a solid at the soil temperature. In this case, the screening level is not limited by C_{sat} because of the reduced possibility of leaching to the water table. If the model estimates a risk-based screening level greater than C_{sat} for a solid in soils, the model will display the final soil concentration as "NOC" or Not of Concern for the vapor intrusion pathway.

In the case of groundwater contamination, the physical state of the contaminant is not an issue in that the contamination has already reached the water table. Because the equilibrium vapor concentration at the source of emissions cannot be higher than that associated with the solubility limit, the vapor concentration is calculated at the solubility limit if the user enters a groundwater concentration greater than the value of S when forward-calculating risk. When reverse-calculating a risk-based groundwater concentration, the model will display the final groundwater concentration as "NOC" for the vapor intrusion pathway if the model calculates a risk-based level greater than or equal to the value of S . It should be noted, however, that if the soil properties or other conditions specified in the DATENTER worksheet are changed, the final risk-based soil or groundwater concentration must be remodeled.

It should also be understood that if a contaminant is labeled "Not of Concern" for the vapor intrusion pathway, all other relevant exposure pathways must be considered for both contaminated soils and groundwater.

3.4 THE CHEMICAL PROPERTIES SHEET (CHEMPROPS)

The chemical properties sheet provides a summary of the chemical and toxicological properties of the chemical selected for analysis. These data are retrieved from the VLOOKUP sheet by CAS number. All data in the chemical properties sheet are protected.

3.5 THE INTERMEDIATE CALCULATIONS SHEET (INTERCALCS)

The intermediate calculations sheet provides solutions to intermediate variables. Review of the values of the intermediate variables may be helpful in an analysis of the cause-and-effect relationships between input values and model results. All data in the intermediate calculations sheet are protected.

3.6 THE LOOKUP TABLES (VLOOKUP)

The VLOOKUP sheet contains two lookup tables from which individual data are retrieved for a number of model calculations. The first table is the Soil Properties Lookup Table. This table contains the average soil water retention curve data of Schaap and Leij (1998) (see Tables 2 and 4) and the mean grain diameter data of Nielson and Rogers (1990) (see Table 3) by SCS soil type. The second table contains all of the chemical and toxicological data for all of the chemicals included with each model.

3.7 ADDING, DELETING, OR REVISING CHEMICALS

Data for any chemical may be edited, new chemicals added, or existing chemicals deleted from the Chemical Properties Lookup Table within the VLOOKUP worksheet. To begin an editing session, the user must unprotect (unseal) the worksheet (the password is "ABC" in capital letters); editing of individual elements or addition and deletion of chemicals may then proceed. Space has been allocated for up to 150 chemicals in the lookup table. Row number 171 is the last row that may be used to add new chemicals. After the editing session is complete, the user must sort all the data in the lookup table (except the column headers) in ascending order by CAS number. After sorting is complete, the worksheet should again be protected (sealed).

SECTION 4

SOIL GAS MODEL APPLICATION

Two new models have been added allowing the user to input measured soil gas concentration and sampling depth data directly into the spreadsheet. These models eliminate the need for theoretical partitioning of a total volume soil concentration or a groundwater concentration into discrete phases. This section provides instructions for using the soil gas models.

4.1 RUNNING THE MODELS

Two models are provided as both MICROSOFT EXCEL and LOTUS 1-2-3 spreadsheets. The screening-level models are entitled SG-SCREEN.xls (EXCEL) and SG-SCREEN.wk4 (1-2-3). The advanced models are SG-ADV.xls and SG-ADV.wk4.

Both the screening-level and advanced models allow the user to calculate steady-state indoor air concentrations and incremental risks from user-defined soil gas concentration data. The models do not allow for reverse-calculation of a risk-based soil or groundwater concentration. As with the soil and groundwater screening-level models, the SG-SCREEN model operates under the assumption that the soil column properties are homogeneous and isotropic from the soil surface to an infinite depth. In addition, the SG-SCREEN model uses the same default values for the building properties as the SL-SCREEN and GW-SCREEN models. Likewise, the SG-ADV model operates under assumptions similar to those of the SL-ADV and GW-ADV models. The advanced model allows the user to specify up to three different soil strata from the bottom of the building floor in contact with the soil to the soil gas sampling depth. Finally, the advanced model allows the user to specify values for all of the model variables.

To run the models, simply open the appropriate file within either MICROSOFT EXCEL or LOTUS 1-2-3. Each model is constructed of the following worksheets:

1. DATENTER (Data Entry Sheet)
2. CHEMPROPS (Chemical Properties Sheet)
3. INTERCALCS (Intermediate Calculations Sheet)
4. RESULTS (Results Sheet)
5. VLOOKUP (Lookup Tables)

Each worksheet follows the form of the worksheets in the soil and groundwater models. See Section 3 for a description of each worksheet.

The DATENTER worksheet of each of the soil gas models is different than those of the soil and groundwater models. Figure 10 shows the DATENTER worksheet of the SG-ADV model. Note that there is no option for running the model to calculate a risk-based media concentration. As with the other models, the user enters the CAS number of the chemical of interest. This automatically retrieves the chemical and toxicological data for that chemical. The CAS number must match one of the 93 chemicals listed in the VLOOKUP worksheet or the message "CAS No. not found" will appear in the "Chemical" box. Next, the user must enter a value for the soil gas concentration of the chemical of interest. The user may enter this value in units of $\bullet \text{ g/m}^3$ or parts-per-million by volume (ppmv). If the soil gas concentration is entered in units of ppmv, the concentration is converted to units of $\bullet \text{ g/m}^3$ by:

$$C_g' = \frac{C_g \times MW}{R \times T_s} \quad (33)$$

where

C_g'	=	Soil gas concentration, $\bullet \text{ g/m}^3$
C_g	=	Soil gas concentration, ppmv
MW	=	Molecular weight, g/mol
R	=	Gas constant (= $8.205 \text{ E-05 atm-m}^3/\text{mol-}^\circ\text{K}$)
T_s	=	System (soil) temperature, $^\circ\text{K}$.

ENTER Chemical CAS No. (numbers only, no dashes)	ENTER Soil gas conc., C_g ($\mu\text{g}/\text{m}^3$)	OR	ENTER Soil gas conc., C_g (ppmv)	Chemical				
71432			2.00E+01	Benzene				
ENTER Depth below grade to bottom of enclosed space floor, L_F (cm)	ENTER Soil gas sampling depth below grade, L_s (cm)	ENTER Average soil temperature, T_s ($^{\circ}\text{C}$)	ENTER Totals must add up to value of L_s (cell C24) Thickness of soil stratum A, h_A (cm)	ENTER Thickness of soil stratum B, (Enter value or 0) h_B (cm)	ENTER Thickness of soil stratum C, (Enter value or 0) h_C (cm)	ENTER Soil stratum A SCS soil type (used to estimate soil vapor permeability)	OR	ENTER User-defined stratum A soil vapor permeability, k_v (cm^2)
200	400	10	200	100	100	L		
ENTER Stratum A soil dry bulk density, ρ_b^A (g/cm^3)	ENTER Stratum A soil total porosity, n^A (unitless)	ENTER Stratum A soil water-filled porosity, θ_w^A (cm^3/cm^3)	ENTER Stratum B soil dry bulk density, ρ_b^B (g/cm^3)	ENTER Stratum B soil total porosity, n^B (unitless)	ENTER Stratum B soil water-filled porosity, θ_w^B (cm^3/cm^3)	ENTER Stratum C soil dry bulk density, ρ_b^C (g/cm^3)	ENTER Stratum C soil total porosity, n^C (unitless)	ENTER Stratum C soil water-filled porosity, θ_w^C (cm^3/cm^3)
1.5	0.43	0.15	1.5	0.43	0.25	1.7	0.38	0.3
ENTER Enclosed space floor thickness, L_{crack} (cm)	ENTER Soil-bldg. pressure differential, ΔP ($\text{g}/\text{cm} \cdot \text{s}^2$)	ENTER Enclosed space floor length, L_B (cm)	ENTER Enclosed space floor width, W_B (cm)	ENTER Enclosed space height, H_B (cm)	ENTER Floor-wall seam crack width, w (cm)	ENTER Indoor air exchange rate, ER (1/h)		
15	40	961	961	488	0.1	0.45		
ENTER Averaging time for carcinogens, AT_C (yrs)	ENTER Averaging time for noncarcinogens, AT_{NC} (yrs)	ENTER Exposure duration, ED (yrs)	ENTER Exposure frequency, EF (days/yr)					
70	30	30	350					

Figure 10. SG-ADV Data Entry Worksheet

In the soil gas models, the steady-state indoor air concentration is calculated by Equation 19 (i.e., $C_{\text{building}} = \alpha C_{\text{source}}$). The value of the vapor concentration at the source of emissions (C_{source}) is assigned the value of the user-defined soil gas concentration. The value of the steady-state attenuation coefficient (α) in Equation 19 is calculated by Equation 13 (see the discussion in Section 2.7). Because no evaluation has been made of the extent of the source of emissions, steady-state conditions (i.e., a non-diminishing source) must be assumed.

The SG-SCREEN model operates under the assumption of homogeneously distributed soil properties and isotropic conditions with respect to soil vapor permeability from the soil surface to an infinite depth. The SG-ADV model, on the other hand, allows the user to specify up to three different soil strata between the building floor in contact with the soil and the soil gas sampling depth. Soil properties within these three strata may be varied allowing for different diffusion resistances to vapor transport (see "Model Variables" in Section 3.2.2 for a discussion of all model variables).

4.2 SOIL GAS SAMPLING

In order to use the soil gas models, soil gas concentrations must be measured at one or more depths below ground surface (bgs). The user is advised to take samples directly under building slabs or basement floors when possible. This can be accomplished by drilling through the floor and sampling through the drilled hole. Alternatively, an angle-boring rig can be used to sample beneath the floor from outside the footprint of the building. When sampling directly beneath the floor is not possible, enough samples adjacent to the structure should be taken to adequately estimate an average concentration based on reasonable spatial and temporal scales.

Soil gas measurements can be made using several techniques, however, active whole-air sampling methods and active or passive sorbent sampling methods are usually employed. Typically, a whole-air sampling method is used whereby a non-reactive sampling probe is inserted into the soil to a prescribed depth. This can be accomplished manually using a "slam bar", a percussion power drill, or the probe can be inserted into the ground using a device such as a Geoprobe®. The Geoprobe® device is attached to the

rear of a specially customized vehicle. In the field, the rear of the vehicle is placed over the sample location and hydraulically raised on its base. The weight of the vehicle is then used to push the sampling probe into the soil. A built-in hammer mechanism allows the probe to be driven to predetermined depths up to fifty (50) feet depending on the type of soil encountered. Soil gas samples can be withdrawn directly from the probe rods or flexible tubing can be connected to the probe tips at depth for sample withdrawal.

Whole-air sampling is typically accomplished using an evacuated Summa or equivalent canister, or by evacuation to a Tedlar bag. Normal operation includes the use of an in-line flow controller and a sintered stainless steel filter to minimize particles becoming entrained in the sample atmosphere. For a 6 liter Summa canister, a normal sampling flow rate for a 24-hr integrated sample might be on the order of 1.5 ml/min, however, higher sampling rates can be used for grab samples. The sampling rate chosen, however, must not be so high as to allow for ambient air leakage between the annulus of the probe and the surrounding soils. Depending on the target compounds, excessive air leakage can dilute the sample, in some cases below the analytical detection limits.

One way to check for leakage is to test an aliquot of the sample gas for either nitrogen or oxygen content before the sample is routed to the canister or Tedlar bag. To test for nitrogen in real- or near real-time requires a portable gas chromatograph/mass spectrometer (GC/MS). A portable oxygen meter, however, can be used to test for sample oxygen content in real-time with a typical accuracy of one-half of one percent. If air leakage is detected by the presence of excessive nitrogen or oxygen, the seal around the sample probe at the soil surface as well as all sampling equipment connections and fittings should be checked. Finally, the flow rate may need to be reduced to decrease or eliminate the air leakage.

The collection and concentration of soil gas contaminants can be greatly effected by the components of the sampling system. It is imperative that materials be used that are inert to the contaminants of concern. Areas of sample collection that need particular attention are:

- The seal at the soil surface around the sample probe
- Use of a probe constructed of stainless steel or other inert material

- Minimization of the use of porous or synthetic materials (i.e., PTFE, rubber or most plastics) which may adsorb soil gas and cause cross-contamination
- Purging of the sample probe and collection system before sampling
- Leak-check of sampling equipment to reduce air infiltration
- Keeping the length of all sample transfer lines as short as possible to minimize condensation of extracted gas in the lines.

The choice of analytical methods for whole-air soil gas sampling depends on the contaminants of concern. Concentrations of volatile organic compounds (VOCs) in the soil gas are typically determined using EPA Method TO-14 or TO-15. In the case of semi-volatile compounds, an active sorbent sampling methodology can be used. In this case, a low volume sampling pump is normally used to withdraw the soil gas which is then routed to a polyurethane foam (PUF) plug. Vapor concentrations of semi-volatile contaminants sorbed to the PUF are then determined using EPA Method TO-10. The active soil gas sampling equipment can be assembled to allow for both canister sampling for volatiles and PUF sampling for semi-volatiles.

Passive sorbent sampling involves burial of solid sorbent sampling devices called cartridges or cassettes to a depth of normally 5 feet or less. The cassettes may be configured with one or more sorbents depending on the list of target analytes, and are typically left in-ground for 72 to 120 hours or longer. During this time period, the vapor-phase soil gas contaminants pass through the cassette and are adsorbed as the soil gas moves towards the soil surface by diffusion and/or convection. Analytical methods for sorbent sampling depends on the target analytes and the sorbent used and may include EPA Method TO-10 or a modified EPA Method TO-1. Vapor-phase concentrations for some solid sorbent sampling systems are determined using the total mass of each contaminant recovered, the time in-ground, the cross-sectional area of the cassette, the diffusivity of the compound in air, and a quasi-empirical adsorption rate constant.

Recent EPA technology verification reports produced by the EPA National Exposure Research Laboratory (EPA 1998, 1998a) concluded, at least for two such systems, that the sorbent methodologies accurately accounted for the presence of most of the soil gas contaminants in the studies. Further, the reports concluded that the sorbent systems showed detection of contaminants at low concentrations not reported using an active whole-air sampling system. For one system, however, it was noted that as the vapor

concentrations reported for the whole-air sampling system increased by 1 to 4 orders-of-magnitude, the associated concentrations reported for the sorbent system increased only marginally. Perhaps the best use of such passive sorbent sampling methods is to help confirm which contaminants are present in the soil gas and not necessarily contaminant concentrations.

An excellent discussion of soil gas measurement methods and limitations can be found in the ASTM Standard Guide for Soil Gas Monitoring in the Vadose Zone D5314-92e1. ASTM Standard Guides are available from the ASTM website at:

<http://www.astm.org>.

In addition, soil gas measurement method summaries can be found in the EPA Standard Operating Procedures for Soil Gas Sampling (SOP No. 2042) developed by the EPA Environmental Response Team (ERT) in Edison, New Jersey. This document can be downloaded from the ERT Compendium of Standard Operating Procedures at the following website:

http://www.ert.org/media_resrcs/media_resrcs.asp.

4.2.1 Data Quality and Data Quality Objectives

The results of soil gas sampling must meet the applicable requirements for data quality and satisfy the data quality objectives (DQOs) of the study for which they are intended. DQOs are qualitative and quantitative statements derived from the DQO process that clarify study objectives, define the appropriate type of data, and specify the tolerable levels of potential decision errors that will be used to support site decisions. DQOs are formulated in the first phase of a sampling project.

In the second phase of the project, a Quality Assurance Project Plan (QAPP) translates these requirements into measurement performance specifications and quality assurance/quality control (QA/QC) procedures to provide the data necessary to satisfy the user's needs. The QAPP is the critical planning document for any environmental data collection operation because it documents how QA and QC activities will be implemented during the life of the project.

Development of the DQOs and the QAPP for soil gas sampling should follow the guidance provided by EPA's Quality Assurance Division of the Office of Research and

Development. Guidance documents concerning the development and integration of the DQOs and the QAPP can be obtained from the EPA website at:

http://epa.gov/ncercqa/qa/qa_docs.html.

In addition to the above guidance, the EPA Regional Office and/or other appropriate regulatory agency should be consulted concerning specific sampling requirements.

4.3 ASSUMPTIONS AND LIMITATIONS

As discussed previously, the soil gas models operate under the assumption of steady-state conditions. This means that enough time has passed for the vapor plume to have reached the building of interest directly above the source of contamination and that the vapor concentrations have reached their maximum values. Depending on the depth at which the soil gas is sampled, diffusion of the soil gas towards the building is a function of the soil properties between the building floor in contact with the soil and the sampling depth. Convection of the soil gas into the structure is a function of the building properties and the effective soil vapor permeability. Assumptions and limitations of the soil gas models are the same as those in Section 5 with the exception of the source vapor concentration that is determined empirically through soil gas sampling.

The user should also recognize the inherent limitations of soil gas sampling. First, the geologic variability of the subsurface may be considerable. This may be especially problematic for shallow soil gas sampling because soil moisture content can vary widely as a function of precipitation events and surface runoff. The soil moisture content has an exponential effect on the rate of vapor diffusion. Transformation processes such as biodegradation can also occur in shallow subsurface soils. In some cases, only a relatively thin stratum of bio-active soil can greatly reduce the emission flux towards the soil surface. Finally, subsurface phase equilibria is a dynamic process resulting in varying vapor-phase concentrations over time at the same sampling location and depth. These factors can result in significant differences in measured soil gas concentrations over relatively small spatial and temporal scales.

For these reasons, the planning phase of the soil gas sampling program should carefully consider the inherent uncertainties in site-specific sampling and analytical data.

In the final analysis, the extent of soil gas sampling is a trade-off between sampling costs and the degree of certainty required in the soil gas concentration data.

SECTION 5

MODEL ASSUMPTIONS AND LIMITATIONS

The following is a discussion of the major assumptions and limitations of the Johnson and Ettinger vapor intrusion model as well as the ancillary models for estimating the vapor concentration at the source of contamination, soil vapor permeability, the rise of the capillary zone, and the effective diffusion coefficient across the capillary zone.

5.1 SOURCE VAPOR CONCENTRATION

As applied in the accompanying spreadsheets, the vapor equilibrium model employed to estimate the vapor concentration at the source of soil contamination is applicable in the limit of "low" concentrations where compounds are sorbed to organic carbon in the soil, dissolved in soil moisture, and present as vapor within the air-filled soil pores (i.e., a three-phase system). The model does not account for a residual phase (e.g., NAPL). If residual phase contaminants are present in the soil column, the user is referred to either the NAPL-SCREEN or NAPL-ADV model, as appropriate.

In the case of contaminated groundwater, the vapor equilibrium model operates under the assumption that the contaminant is present at levels below the water solubility limit. If the user-defined soil concentration is greater than the soil saturation concentration (C_{sat}) or if the groundwater concentration is greater than the solubility limit (S), the equilibrium vapor concentration will be calculated at the value of C_{sat} or S as appropriate.

The user is also reminded that when estimating a risk-based soil concentration, the model will compare the calculated soil concentration with the soil saturation concentration above which a residual phase is likely to occur. The soil saturation concentration (C_{sat}) is calculated as in U.S. EPA (1996a and b). If the risk-based concentration is greater than the saturation concentration and the contaminant is a liquid or gas at the soil temperature, the final soil concentration will be set equal to the soil saturation concentration. This tends to eliminate the possibility of allowing a liquid residual phase to exist within the soil column,

which may leach to the water table. If the risk-based soil concentration is greater than C_{sat} and the contaminant is a solid, the contaminant is not of concern for the vapor intrusion pathway.

Likewise, the groundwater models will compare the calculated risk-based groundwater concentration to the aqueous solubility limit of the compound. If the risk-based groundwater concentration is greater than the solubility limit, the contaminant is not of concern for the vapor intrusion pathway.

Finally, it should be recognized that the procedures used to estimate both the soil saturation concentration and the aqueous solubility limit do not consider the effects of multiple contaminants. The estimated values, therefore, may be artificially high such that a residual phase may actually exist at somewhat lower concentrations.

5.2 SOIL VAPOR PERMEABILITY

The procedures used to estimate the soil vapor permeability of the soil stratum in contact with the building floor and walls assumes isotropic soils and steady-state soil moisture content. In addition, the calculations do not account for preferential vapor pathways due to soil fractures, vegetation root pathways, or the effects of a gravel layer below the floor slab or backfill which may act to increase the vapor permeability with respect to in situ soils.

If in situ pneumatic tests are used to measure site vapor permeability, care must be taken to ensure adequate sampling to reduce the possibility of missing important soil structure effects due to anisotropy.

Single point in situ pneumatic tests are typically conducted by measuring the pressure in a probe as a metered flow of air is passed through the probe and into the soil. Garbesi et al. (1996), however, demonstrated that soil vapor permeability increases with the sampling length scale. Using a dual-probe dynamic pressure sampling apparatus, Garbesi et al. (1996) demonstrated that the average soil vapor permeability typically increases up to a constant value as the distance between the source probe and detector probe increases. On a length scale typical of a house (3 to 10 m) use of the dual-probe sampling technique found that the soil permeability was approximately 10 to 20 times

higher than that measured by the single point method. Although arguably the most accurate means of determining in situ soil vapor permeability, the techniques of Garbesi et al. (1996) are complex and require specialized equipment.

Another method for determining the intrinsic permeability of soil is to conduct empirical measurements of the saturated hydraulic conductivity (K_s). These data are then input into Equation 26. The resulting value of k_i is then multiplied by the relative air permeability (k_{rg}) calculated by Equation 27 to yield the effective air permeability of the soil.

5.3 RISE OF AND DIFFUSION ACROSS THE CAPILLARY ZONE

Estimation of the rise of the capillary zone is based on the equation for the rise of a liquid in a capillary tube. The procedure assumes that the interstitial space between the soil particles is equivalent to the capillary tube diameter and that the resulting rise of water occurs under steady-state soil column drainage conditions. In actuality, the height of the capillary zone is uneven or fingered due to the variation in the actual in situ particle size distribution. In addition, the groundwater models do not account for the episodic rise and fall of the water table or the capillary zone due to aquifer recharge and discharge. As constructed, the groundwater models do not allow the top of the capillary zone to be above the bottom of the building floor in contact with the soil. The user should be aware, however, that in reality the top of the capillary zone may rise to levels above the floor in some cases.

Diffusion across the capillary zone is estimated based on lumping vapor and aqueous-phase diffusion together within the calculation of the effective diffusion coefficient.

To allow for vapor-phase diffusion within the capillary zone, the air-filled soil pores must be connected. In reality, the capillary zone may be comprised of a tension-saturated zone immediately above the water table and the deep portion of the vadose zone within which the soil water content is a strongly dependent on the pressure head. Diffusion across the tension-saturated zone is dominated by liquid-phase diffusion which is typically four orders of magnitude less than vapor-phase diffusion. Therefore, a large concentration gradient

may exist between the top of the water table and the top of the tension-saturated zone (McCarthy and Johnson, 1993).

Lumping vapor and aqueous-phase diffusion together is a less intensive, although less rigorous, method for estimating the effective diffusion coefficient. The result is typically a higher effective diffusion coefficient relative to separate solutions for aqueous diffusion across the tension-saturated zone and both vapor and aqueous diffusion across the unsaturated portion of the vadose zone.

To minimize the possible over estimation of the effective diffusion coefficient, the soil air-filled porosity within the capillary zone is estimated based on the air-entry pressure head, which corresponds with the water-filled porosity at which the interstitial air-filled pores first become connected. The user should be aware that this procedure is inherently conservative if a significant concentration gradient exists across the tension-saturated zone. This conservatism may be somewhat offset in that the model does not consider any episodic rise in the level of the water table. During such events, water which had previously been part of the saturated zone (and hence contain higher contaminant concentrations) is redistributed in the vadose zone resulting in temporary elevations in soil gas concentrations.

5.4 DIFFUSIVE AND CONVECTIVE TRANSPORT INTO THE STRUCTURE

The following is a discussion of the major assumptions and limitations of the Johnson and Ettinger model for diffusive and convective vapor transport into buildings.

The model assumes that all vapors from underlying soils will enter the building through gaps and openings in the walls, floor and foundation. This implies that a constant pressure field is generated between the interior spaces and the soil surface and that the vapors are intercepted within the pressure field and transported into the building. This assumption is inherently conservative in that it neglects periods of near zero pressure differential (e.g., during mild weather when windows are left open).

As with the estimation procedure for soil vapor permeability, the model assumes isotropic soils in the horizontal direction; vertical anisotropy is accounted for by a series of isotropic soil strata above the top of contamination. Soil properties within the zone of soil

contamination are assumed to be identical to those of the soil stratum directly above the contamination and extend downward to an infinite depth. Solute transport by convection (e.g., water infiltration) and by mechanical dispersion are neglected. Transformation processes (e.g., biodegradation, hydrolysis, etc.) are also neglected.

An empirical field study (Fitzpatrick and Fitzgerald, 1997) indicated that the model may be overly conservative for nonchlorinated species (e.g., benzene, toluene, ethylbenzene and xylene) but in some cases, may underpredict indoor concentrations for chlorinated species. The authors contribute the likely cause for this discrepancy to the significant biodegradation of the nonchlorinated compounds.

The Johnson and Ettinger model treats the entire building as a single chamber with instantaneous and homogeneous vapor dispersion. It therefore neglects contaminant sinks and the room-to-room variation in vapor concentration due to unbalanced mechanical and/or natural ventilation.

Finally, convective vapor flow from the soil matrix into the building is represented as an idealized cylinder buried below grade. This cylinder represents the total area of the structure below the soil surface (walls and floor). The total crack or gap area is assumed to be a fixed fraction of this area. Because of the presence of basement walls, the actual vapor entry rate is expected to be 50 to 100 percent of that provided by the idealized geometry (Johnson and Ettinger, 1991).

SECTION 6

VALUES FOR INPUT VARIABLES

The Johnson and Ettinger model as constructed within the accompanying spreadsheets requires a range of input variables depending on whether a screening-level or advanced model is chosen.

Table 5 provides a list of all major input variables, the range of practical values for each variable, the default value for each variable, and the relative model sensitivity and uncertainty of each variable. Table 5 also includes references for each value or range of values.

Table 6 indicates the results of an increase in the value of each input parameter. The results are shown as either an increase or a decrease in the building concentration (C_{building}) of the pollutant. An increase in the building concentration will result in an increase in the risk when forward-calculating from an initial soil or groundwater concentration. When reverse-calculating to a risk-based “acceptable” soil or groundwater concentration, an increase in the hypothetical unit building concentration will result in a lower “acceptable” soil or groundwater concentration.

TABLE 5. RANGE OF VALUES, SENSITIVITY, AND UNCERTAINTY OF MODEL INPUT PARAMETERS

Input parameter	Practical range of values	Default value	Relative model sensitivity	Relative uncertainty
Soil water-filled porosity (θ_w)	0.02 - 0.43 cm ³ /cm ^{3a}	0.30 cm ³ /cm ^{3a}	High	High
Soil vapor permeability (k_v)	10 ⁻⁶ - 10 ⁻¹² cm ^{2b,c}	10 ⁻⁸ cm ^{2d}	High	High
Soil-building pressure differential (ΔP)	0 - 20 Pa ^e	4 Pa ^f	High	High
Media initial concentration (C_R , C_w)	User-defined	NA	High	Moderate
Depth to bottom of soil contamination (L_b)	User-defined	NA	High	Moderate
Depth to top of contamination (L_T)	User-defined	NA	High	Low
Floor-wall seam gap (w)	0.05 - 1.0 cm ^g	0.1 cm ^g	Moderate	High
Soil organic carbon fraction (f_{oc})	0.001 - 0.006 ^a	0.002 ^a	Moderate	Moderate
Building air exchange rate (ER)	0.18 - 1.26 (h ⁻¹) ^g	0.45 (h ⁻¹) ^{g,h}	Moderate	Moderate
Building volume	147 - 672 m ³ⁱ	451 m ³ⁱ	Moderate	Low
Soil total porosity (n)	0.34 - 0.53 cm ³ /cm ^{3a}	0.43 cm ³ /cm ^{3a}	Moderate	Low
Soil dry bulk density (ρ_b)	1.25 - 1.75 g/cm ^{3a}	1.5 g/cm ^{3a}	Low	Low

^aU.S. EPA (1996a and b).

^bJohnson and Ettinger (1991).

^cNazaroff (1988).

^dBased on transition point between diffusion and convection dominated transport from Johnson and Ettinger (1991).

^eEaton and Scott (1984); Loureiro et al. (1990).

^fLoureiro et al. (1990); Grimsrud et al. (1983).

^gKoontz and Rector (1995).

^hParker et al. (1990).

ⁱU.S. DOE (1995).

TABLE 6. EFFECT ON BUILDING CONCENTRATION FROM AN INCREASE IN INPUT PARAMETER VALUES

Input parameter	Change in parameter value	Effect on building concentration
Soil water-filled porosity (θ_w)	Increase	Decrease
Soil vapor permeability (k_v)	Increase	Increase
Soil-building pressure differential (ΔP)	Increase	Increase
Media initial concentration (C_R, C_w) ^a	Increase	Increase
Depth to bottom of soil contamination (L_b) ^b	Increase	Increase
Depth to top of contamination (L_T)	Increase	Decrease
Floor-wall seam gap (w)	Increase	Increase
Soil organic carbon fraction (f_{oc})	Increase	Decrease
Building air exchange rate (ER)	Increase	Decrease
Building volume ^c	Increase	Decrease
Soil total porosity (n)	Increase	Increase
Soil dry bulk density (ρ_b)	Increase	Decrease

^aThis parameter is applicable only when forward-calculating risk.

^bApplicable only to advanced model for soil contamination.

^cUsed with building air exchange rate to calculate building ventilation rate.

SECTION 7

INTERPRETATION OF RESULTS

The models described herein are theoretical approximations of complex physical and chemical processes and as such should not be used in a deterministic fashion (i.e., to generate a single outcome). At the least, a range of outcomes should be explored focusing on the most sensitive model input variables. In general, using the default values for input variables will result in higher indoor air concentrations and thus higher incremental risks or lower risk-based media concentrations. With a realistic range of outcomes, the risk manager may assess the uncertainty in the model predictions.

From a conceptual point of view, the Johnson and Ettinger model provides a theoretical description of the processes involved in vapor intrusion from subsurface soils or groundwater into indoor structures. A combination of modeling and sampling methods is also possible to reduce the uncertainty of the calculated indoor air concentrations. Typically this involves field methods for measuring soil gas very near or below an actual structure. It should be understood, however, that soil gas sampling results outside the footprint of the building may or may not be representative of the soil gas concentrations directly below the structure. For solid building floors in contact with the soil (e.g., concrete slabs), the soil gas directly beneath the floor may be considerably higher than that adjacent to the structure. This is typically due to a vapor pooling effect underneath the near impermeable floor. Once a representative average concentration is determined, all vapor directly below the areal extent of the building is presumed to enter the structure. The soil gas concentration, along with the building ventilation rate and the soil gas flow rate into the building, will determine the indoor concentration. When using the soil gas models, it must be remembered that no analysis has been made concerning the source of contamination. Therefore, the calculated indoor concentration is assumed to be steady-state. The procedures described in API (1998) can be used to calibrate the diffusion transport

considerations of the Johnson and Ettinger model as well as for calibrating the model for transformation processes (e.g., biodegradation). The reader is also referred to U.S. EPA (1992) for a more detailed discussion of applying soil gas measurements to indoor vapor intrusion.

Finally, calibration and verification of the model has been limited to radon studies due to the paucity of suitable data. Research is needed to provide spatially and temporally correlated measurements during different seasons, at different locations, with different buildings, and over a range of different contaminants such that the accuracy of the model may be determined.

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Evaluation of the Johnson and Ettinger Model for Prediction of Indoor Air Quality

by Ian Hers, Reidar Zapf-Gilje, Paul C. Johnson, and Loretta Li

Abstract

Screening level models are now commonly used to estimate vapor intrusion for subsurface volatile organic compounds (VOCs). Significant uncertainty is associated with processes and models and, to date, there has been only limited field-based evaluation of models for this pathway. To address these limitations, a comprehensive evaluation of the Johnson and Ettinger (J&E) model is provided through sensitivity analysis, comparisons of model-predicted to measured vapor intrusion for 11 petroleum hydrocarbon and chlorinated solvent sites, and review of radon and flux chamber studies. Significant intrusion was measured at five of 12 sites with measured vapor attenuation ratios (α_m 's) (indoor air/source vapor) ranging from $\sim 1 \times 10^{-6}$ to 1×10^{-4} . Higher attenuation ratios were measured for studies using radon, inert tracers, and flux chambers; however, these ratios are conservative owing to boundary conditions and tracer properties that are different than those at most VOC-contaminated sites. Reasonable predictions were obtained using the J&E model with comparisons indicating that model-predicted vapor attenuation ratios (α_p 's) were on the same order, or less than the α_m 's. For several sites, the α_m were approximately two orders of magnitude less than the α_p 's indicating that the J&E model is conservative in these cases. The model comparisons highlight the importance in using appropriate input parameters for the J&E model. The regulatory implications associated with use of the J&E model to derive screening criteria are also discussed.

Introduction

The use of models to predict indoor air quality associated with volatile organic compound (VOC) contamination in soil and ground water is now commonplace (ASTM 1995; Johnson et al. 1998; Hers et al. 2002). Screening models typically used for this pathway are the Johnson and Ettinger (1991) model (henceforth referred to as the J&E model), or variants thereof. Processes controlling the intrusion of VOC vapors into buildings are not well understood, the accuracy of the J&E model is uncertain, and there have been only limited comparisons of model predictions to field data. There are also substantial differences in the way in which the J&E model is used for regulatory purposes.

To address these limitations, this paper presents a comprehensive evaluation of the J&E model based on theoretical considerations and field data from petroleum hydrocarbon and chlorinated solvent sites, and radon and flux chamber studies. Data sources are published studies, consultant or agency reports, and a field-based research program conducted by the authors. Included in the data sets analyzed are several recent groundbreaking investigations at chlorinated solvent sites.

The paper begins with an analysis of methods for estimating input parameters for the J&E model and their effect on model sensitivity and uncertainty. This analysis provides the needed context for the methods employed to interpret the field data used for this study. It is also important because it is

essential that model attributes and potential limitations be understood before using field data to evaluate the predictive capabilities of a model. Field-based methods for the evaluation of vapor attenuation ratio (α), defined as the indoor air concentration divided by the source vapor concentration, are evaluated next. The primary focus is measured vapor attenuation ratios (α_m) from 11 sites with petroleum hydrocarbon and chlorinated solvent contamination. Information from tracer studies using radon or an injected tracer such as sulphur hexafluoride (SF_6), and flux chamber studies are also reviewed. The measured α_m from field studies are compared to model-predicted vapor attenuation ratios (α_p) using the J&E model. Trends in the data are qualitatively evaluated and possible factors affecting vapor intrusion are considered. The paper also comments on the use of the J&E model to derive regulatory screening criteria.

J&E Model Input Parameters, Sensitivity, and Uncertainty

The basic form of the J&E model couples one-dimensional steady-state diffusion through soil, and diffusion and advection through a building envelope (i.e., foundation). A simple "box" model, which assumes uniform and instantaneous mixing of chemicals within the building enclosure, is used to estimate the indoor air concentration. Model sensitivity and uncertainty analysis and input needed for comparisons of

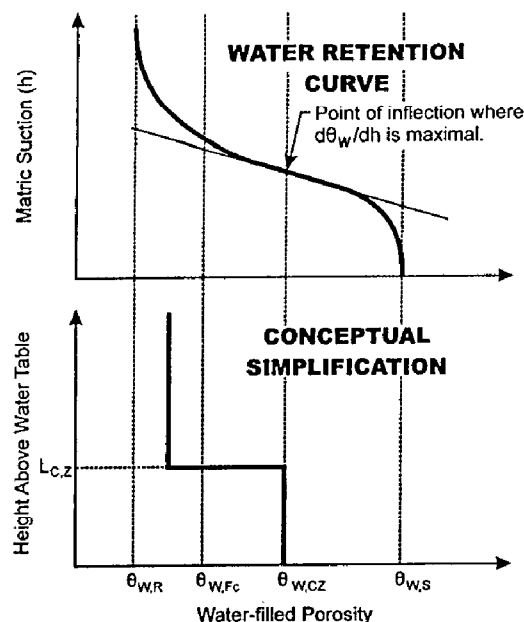


Figure 1. Conceptual simplification of water retention curve for purposes of estimating moisture contents and capillary rise ($\theta_{w,r}$, $\theta_{w,fc}$, $\theta_{w,cz}$, $\theta_{w,s}$ are the residual, field capacity, capillary zone, and saturated water contents).

model predictions to field data all require estimation of effective diffusion coefficient and soil gas advection rate. Because the available data varied, different methods were used to estimate these input parameters and interpret field data. The estimation methods subsequently used in this paper are discussed in the following sections.

Estimation of Effective Diffusion Coefficient (Air-Filled and Total Porosity)

The J&E model uses the Millington and Quirk (1961) relationship to estimate the effective diffusion coefficient (D_T^{eff}), as follows:

$$D_T^{eff} = (\theta_a^{(10/3)} / \theta^2) * D_{air} + 1/H' * (\theta_w^{(10/3)} / \theta^2) * D_{water}$$

where θ_a , θ_w , and θ are the air-filled, water-filled, and total porosity; D_{air} and D_{water} are free-air and free-water diffusion coefficients (L^2T^{-1}); and H' is the dimensionless Henry's law constant.

A common method for estimating air-filled and total porosity directly uses the measured soil moisture content and bulk density. A potential disadvantage is that soil disturbance during sampling can lead to inaccurate moisture, density, and hence, porosity estimates. Samples obtained adjacent to buildings may not be representative of conditions below buildings owing to the drying of soil that can occur.

A second method involves the use of the van Genuchten (VG) model (van Genuchten 1980) to predict the water retention parameters for U.S. Soil Conservation Service (SCS) soil types, based on VG model curve-fit parameters computed by Schaap and Leij (1998) (Simplified VG method). This method, developed by Environmental Quality Management Inc. (EQM 2000), is incorporated in U.S. EPA guidance for this

pathway. The VG model parameters are, in turn, used to develop a simplified step function for water-filled porosity (Figure 1). The capillary zone ($\theta_{w,cz}$) water-filled porosity is equal to the moisture content at the inflection point in the water retention curve where $d\theta_w/dh$ is maximal, as suggested by Waitz et al. (1996) (where θ_w and h equal the water-filled porosity and matric suction, respectively). Vapor-phase diffusion becomes negligible once the water-filled porosity exceeds the $\theta_{w,cz}$. The height of the capillary zone is estimated using an equation for capillary rise in a tube (Fetter 1994), and mean particle size for the SCS soil textural classifications (Nielson and Rogers 1990). The water-filled porosity above the capillary zone is user defined; we suggest a practical range below a building is between the residual water content and field capacity.

The simplified VG model likely predicts lower than actual water-filled porosity in soil, for the capillary transition zone (Figure 1). Because diffusion rates are much higher in air than water, this simplification likely results in conservative (high) diffusion estimates through the capillary transition zone. However, this conservatism may be counterbalanced by nonrepresentative assumptions for the ground water contamination source. The common paradigm for prediction of cross-media VOC transport is that dissolved chemicals are present below a static water table, and that transport through the capillary transition zone is limited to vapor- and aqueous-phase diffusion. In reality, there will be some lateral ground water flow and dispersive mixing of chemicals in the tension-saturated zone, and vertical movement of chemicals as a result of water-table fluctuations. There is limited information on VOC migration in the capillary transition zone. One study, involving a large chamber, showed that the pore-water concentrations in the tension-saturated zone were similar to those below the water table, and showed a sharp decline in concentrations near the top of the tension-saturated zone (McCarthy and Johnson 1993). The implication is that a more representative top boundary for dissolved ground water contaminants may be some distance above the water table.

Estimation of Soil Gas Advection Rate (Q_{soil})

The method often used with the J&E model for estimating the soil gas advection rate (Q_{soil}) through the building envelope is an analytical solution for two-dimensional soil gas flow to a small horizontal drain (Nazaroff 1992). This model is used to simulate gas flow to an edge crack located at the perimeter of a building (perimeter crack model). The Q_{soil} (L^3T^{-1}) is estimated as follows:

$$Q_{soil} = \frac{2 \pi k_a \Delta P X_{crack}}{\mu \ln \left(\frac{2 z_{crack}}{r_{crack}} \right)} \quad (2)$$

where k_a is the soil-air permeability (L^2), ΔP is the pressure difference between the building and ambient air, X_{crack} is the perimeter crack length (L), μ is the gas viscosity ($ML^{-1}T^{-1}$), z_{crack} is the depth to edge crack (L), and r_{crack} is the crack radius (L). The ratio of cracks to total subsurface foundation area (i.e., base and walls) (η) can be expressed as

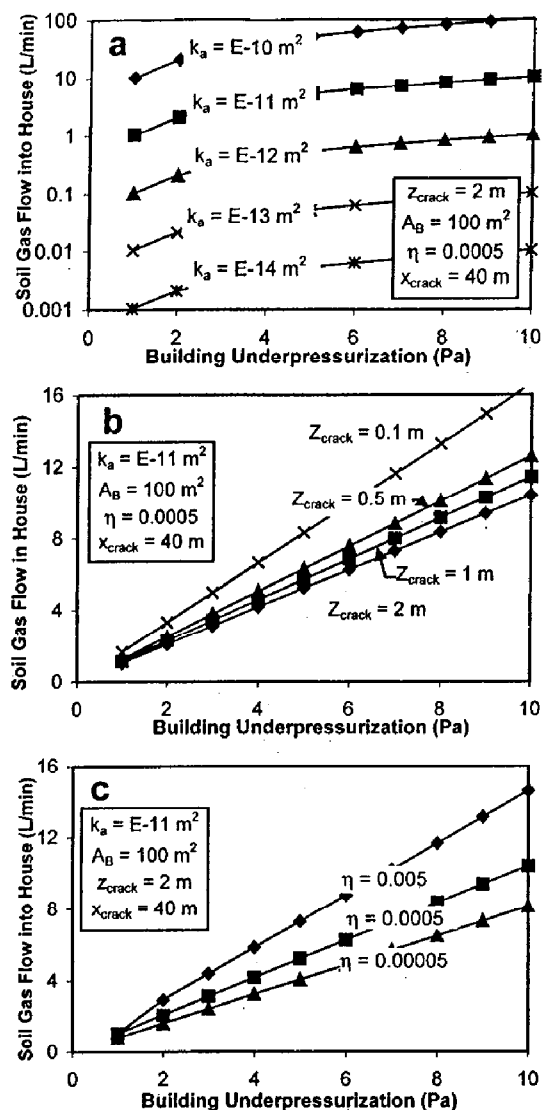


Figure 2. Sensitivity of soil gas flow to perimeter crack model (used in J&E model) to (a) soil-air permeability (k_a), (b) depth to perimeter crack (z_{crack}), and (c) crack ratio (η). x_{crack} = perimeter crack length, A_B = subsurface foundation area.

$$\eta = \frac{r_{\text{crack}} x_{\text{crack}}}{A_B} \quad (3)$$

where A_B is the subsurface foundation area (L^2). The perimeter crack model accounts for both soil gas flow through soil and the foundation, but is most sensitive to the soil-air permeability based on the analysis presented in Figure 2. For the range of values chosen for k_a , η , ΔP , and z_{crack} , by far the greatest variation is obtained for k_a with the predicted Q_{soil} ranging between ~ 0.001 and 100 L/min .

One method of estimating soil-air permeability is to use published values for saturated hydraulic conductivity and water retention parameters for a particular soil type (EQM 2000). This method involves the following steps: (1) obtain saturated hydraulic conductivity for soil texture type (Schaap and Leij 1998); (2) estimate intrinsic permeability from saturated hydraulic conductivity; (3) estimate effective total fluid saturation at field capacity;

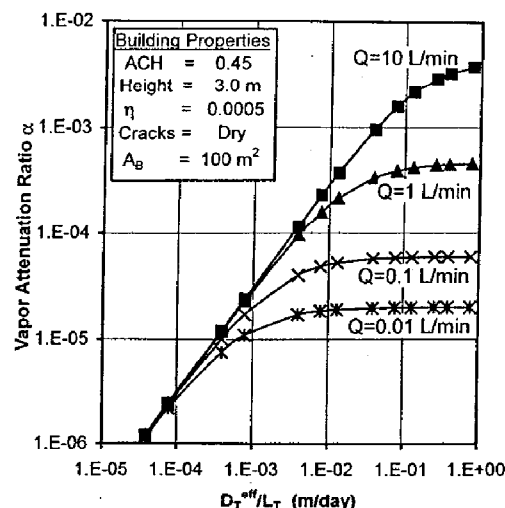


Figure 3. Sensitivity of vapor attenuation ratio (benzene) to soil-gas flow rate (Q) into building using perimeter crack model with dry dust-filled concrete cracks with total porosity = 0.3 Height = building height, $Q = Q_{\text{soil}}$ ACH = air exchanges per hour (other symbols previously defined).

(4) estimate relative air permeability using the relationship proposed by Parker et al. (1987); and (5) calculate effective soil-air permeability (relative air permeability multiplied by intrinsic permeability). The soil-air permeability can also be measured in the field (Garbesi and Sextro 1995; Hers and Zapf-Gilje 1998); however, this type of testing is rarely performed.

The Q_{soil} can also be estimated from a tracer test mass balance. When soil-gas advection is the primary mechanism for tracer intrusion into a building, the Q_{soil} can be estimated by measuring the concentrations of a chemical tracer in indoor air, outdoor air, and in soil vapor below a building, and measuring the building ventilation rate (Hers et al. 2002; Fischer et al. 1996; Garbesi et al. 1993; Rezvan et al. 1991; Garbesi and Sextro, 1989). The Q_{soil} values measured using this technique are compared to predicted rates using the perimeter crack model, for sites with coarse-grained soils (Table 1). The perimeter crack model predictions are both higher and lower than the measured values, but overall are within one order of magnitude of the measured values. Although the Q_{soil} predicted by models and measured using field tracer tests are uncertain, the results suggest that a "typical" range for houses on coarse-grained soils is on the order of 1 to 10 L/min.

J&E Model Sensitivity for Key Input Parameters

The sensitivity of the benzene α_p predicted by the J&E model is evaluated as a function of soil gas flow (Q_{soil}), the effective diffusion coefficient (D_T^{eff}), and contamination depth (L_T) (Figure 3). The D_T^{eff}/L_T ratio captures the influence of soil properties and depth to contamination source on α_p . For BTEX and most chlorinated solvent compounds, chemical-specific variation in the D_T^{eff}/L_T ratio is not significant because the free-air diffusion coefficients vary by only a factor of two, and the Henry's law constants vary by a factor of 10 (D_T^{eff}/L_T is less sensitive to H' than D_{air}). Because the effective diffusion coefficient is calculated using the Millington and Quirk (1961) relationship, the soil properties of relevance are the air-filled and total porosity. A high D_T^{eff}/L_T ratio is asso-

Table 1
Comparison of Measured and Model-Predicted Soil Gas Flow Rates Into Buildings

Site	Foundation Type	ΔP (Pa)	Subsurface Foundation Area (m ²)	Crack Ratio η	Depth to Perimeter Crack (m)	$k_{\text{soil-air}}$ (Darcy)	Soil Gas Flow Rates	
							Measured Tracer (L/min)	Predicted PCM (L/min)
Chatterton Site (Hers et al. 2000)	Slab-on-grade	30	57	0.00033	0.3	10	2.7	29
	Slab-on-grade	10	57	0.00033	0.3	10	4.2	9.6
	Slab-on-grade	10	57	0.0001	0.3	10	2.9	8.2
Alameda Site Fischer et al. (1996)	Slab-on-grade	3	50	0.0001	0.2	10	1.4	2.4
Central California Site Garbesi & Sextro (1989)	Filled hollow block basement w/coating	30	128	0.0001	2.5	3	67	8.3
Ben Lomond Garbesi et al. (1993)	Experimental basement	10	26	0.00075	1.8	6	9.7	2.3
Spokane Valley Houses Revzan et al. (1991)	Poured concrete basements	5	220	0.0001	2	200	102	110

Notes: Bold print values assumed, all other values measured, ΔP = building underpressurization, PCM = Perimeter Crack model.

Table 2
Qualitative Summary of Sensitive Parameters for the J&E Model

	Building Depressurized (Advection and Diffusion)	Building Not Depressurized (Diffusion Only)
High D_T^{eff}/L_T (shallow and/or dry soil)	Q_{soil} (advection controlled)	Building foundation cracks
Moderate D_T^{eff}/L_T	Q_{soil} and moisture content (MC)	Building foundation cracks and MC
Low D_T^{eff}/L_T (deep and/or wet soil)	Moisture content (diffusion controlled)	Moisture content (MC)

Note: Indoor air concentrations are directly proportional to source concentrations, building mixing height and ventilation rate.

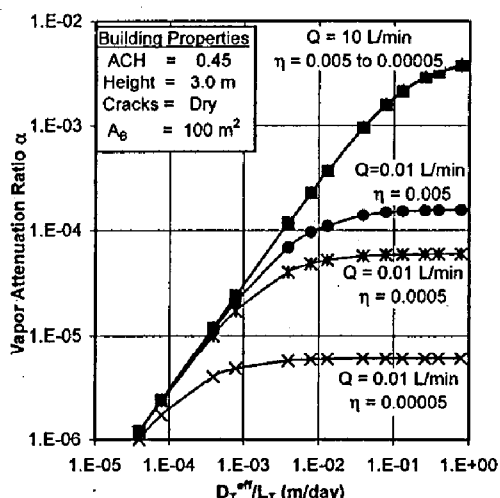


Figure 4. Sensitivity of vapor attenuation ratio (benzene) to soil-gas flow rate (Q) using perimeter crack model and foundation crack ratio (η) (other symbols previously defined).

ciated with dry soils and/or shallow contamination, whereas a low D_T^{eff}/L_T ratio is associated with wet soils and/or deep contamination. Based on the analysis in the sections that follow, sensitive parameters for the J&E model are also qualitatively summarized in Table 2.

Sensitivity of α_p to Q_{soil}

For sensitivity analysis purposes, a Q_{soil} range of 0.01 to 10 L/min was chosen because it is considered representative of most houses or small buildings. The results indicate that Q_{soil} begins to have a significant influence on α_p when D_T^{eff}/L_T values are moderate to relatively large ($> \sim 0.001$ m/day) (Figure 3). The J&E model is described to be advection controlled for this scenario. When D_T^{eff}/L_T is relatively small ($< \sim 0.001$ m/day), α_p is not sensitive to Q_{soil} . The J&E model is described to be diffusion controlled for this scenario. The D_T^{eff}/L_T for case studies subsequently evaluated in this paper ranged from ~ 0.002 to 0.1 m/day. For these D_T^{eff}/L_T values, the maximum error in prediction caused by a four order of magnitude variation in Q_{soil} ranges from 3X to 100X.

Sensitivity of α_p to Crack Ratio

The influence of crack ratio (η) on α_p was evaluated for two different Q_{soil} values (Figure 4). For $Q_{\text{soil}} = 10$ L/min, α_p is not sensitive to η . When $Q_{\text{soil}} = 0.01$ L/min, a two order of magnitude change in η causes up to 25X change in α_p . The sensitivity of α_p to η increases as Q_{soil} decreases, with sensitivity highest for the diffusion-only case (i.e., $Q_{\text{soil}} = 0$). The crack ratio is of little importance for smaller D_T^{eff}/L_T or $Q_{\text{soil}} > \sim 1$ L/min, which means that for the majority of sites crack ratio will not be important.

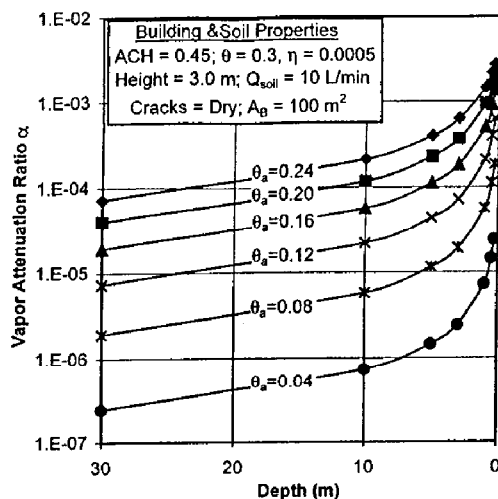


Figure 5. Sensitivity of vapor attenuation ratio (benzene) to water-filled porosity (θ_a). Other symbols previously defined.

Sensitivity of α_p to Air-Filled Porosity (Moisture Content)

The effect of air-filled porosity and depth to contamination was evaluated for a soil with moisture contents ranging from 3.6% to 15.6% (dry weight) and a constant total porosity of 0.3 (Figure 5). This variation in moisture content is potentially representative of the difference between a dry soil below a building compared to a wet soil within the capillary transition zone. The corresponding air-filled porosities are between 0.04 and 0.26. A Q_{soil} value of 10 L/min was assumed. For a constant depth to contamination, a 4X change in moisture content causes approximately or more than two orders of magnitude change in α_p . For a constant moisture content, α_p becomes sensitive to depth to contamination, at shallow depths. It is clear that soil layers with high moisture content will have a significant effect on the diffusive flux and vapor intrusion.

J&E Model Uncertainty for Range of Values

Vapor attenuation ratios predicted by the J&E model are provided for a range of soil gas advection rates and building properties, as a function of D_T^{eff}/L_T (Figure 6). For illustrative purposes, upper and lower soil-gas advection rates were estimated for four U.S. SCS soil textures (sand, loamy sand, sandy loam, and silt) using published values for saturated hydraulic conductivity and the perimeter crack model. The soil type only applies to soil immediately adjacent to the building, because the radius-of-influence for soil-gas advection is relatively limited. The estimated Q_{soil} values are highly uncertain; however, we note that the predicted values for sand (1 to 10 L/min) are consistent with the results of tracer tests for coarse-grained soils. The uncertainty in Q_{soil} increases for finer-grained soils because the influence of permeable soil layers and preferential pathways (e.g., utility backfill) becomes more important. It is suggested that the Q_{soil} for sand be used when near the foundation soil is not well characterized.

The building properties input to the model are the crack ratio, dust-filled crack moisture content, building height, building air exchanges, and building foundation size. The upper and lower building properties given are subjectively considered to represent the range of values that would be encountered at most

sites, based on available information and the author's experience (Hers and Zapf-Gilje 1998; Hers et al. 2001). The sub-surface foundation area is for a house with a shallow basement or slab-on-grade foundation. Slightly lower α_p 's would be predicted for a deep basement with larger foundation area.

The graphs in Figure 6 illustrate the effect of variation in Q_{soil} and building properties on vapor attenuation ratio, but do not address uncertainty in D_T^{eff}/L_T , which is primarily caused by soil moisture content. To gain insight into uncertainty in model predictions owing to moisture content, a possible range in D_T^{eff}/L_T was evaluated for two hypothetical scenarios. The first scenario (Site 1) assumes a shallow soil vapor source (1.5 m depth) situated well above the water table. The second scenario (Site 2) assumes a relatively deep water table (6 m depth) and contamination that is limited to a dissolved ground water plume. Both sites were assumed to have uniform SCS loamy sand soil. The approach taken was to first obtain a plausible best estimate, and upper and lower range for D_T^{eff}/L_T . For Site 1, a constant air-filled porosity halfway between the residual water content and field capacity was assumed. For Site 2, the simplified VG method was used to estimate the air-filled and total porosity for the capillary zone. As shown in Table 3, the resulting porosities are expressed as relative water saturation values where $S = \theta_w/\theta$ and $\theta_a = \theta(1-S)$. The reason for using relative saturation values in the uncertainty analysis is that the air-filled and total porosity are expected to be strongly correlated. Therefore, uncertainty would be overestimated if these parameters are allowed to vary independently. This is prevented through the use of the relative saturation values. The uncertainty ranges given for total porosity and relative saturation are considered reasonable values for a well-characterized site.

Using the best estimate values and uncertainty ranges, the best estimate, lower and upper ranges are provided for the normalized effective diffusion coefficient (D_T^{eff}/L_T) (Table 3 and Figure 6). For Site 1, the upper and lower D_T^{eff}/L_T values vary by a factor of 2.4. For Site 2, the uncertainty is greater (factor of 23) because the sensitivity of D_T^{eff}/L_T to air-filled porosity within the capillary zone is high because moisture content is also high.

The overall uncertainty in the vapor attenuation ratio will be dependent on the available data. If there is information only on the contamination depth, the range in α_p can vary three to four orders of magnitude. When information on soil properties is also available, the uncertainty in D_T^{eff}/L_T and Q_{soil} is reduced resulting in α_p that vary over two orders of magnitude (Figure 6). When good quality site-specific data is available for both soil properties (e.g., moisture content) and building properties (e.g., ventilation rate, mixing height), it may be possible to reduce the uncertainty in α_p to approximately one order of magnitude.

Field-Based Methods for Evaluation of Vapor Intrusion

Three field-based approaches or methods are used to evaluate vapor intrusion: the indoor VOC method, the tracer method, and the flux chamber method. The *indoor VOC method* involves measurement of VOC concentrations in indoor air and at the contamination source. The α_m will vary depending on the contamination scenario. For sites with dissolved ground water plumes, the α_m is calculated using a

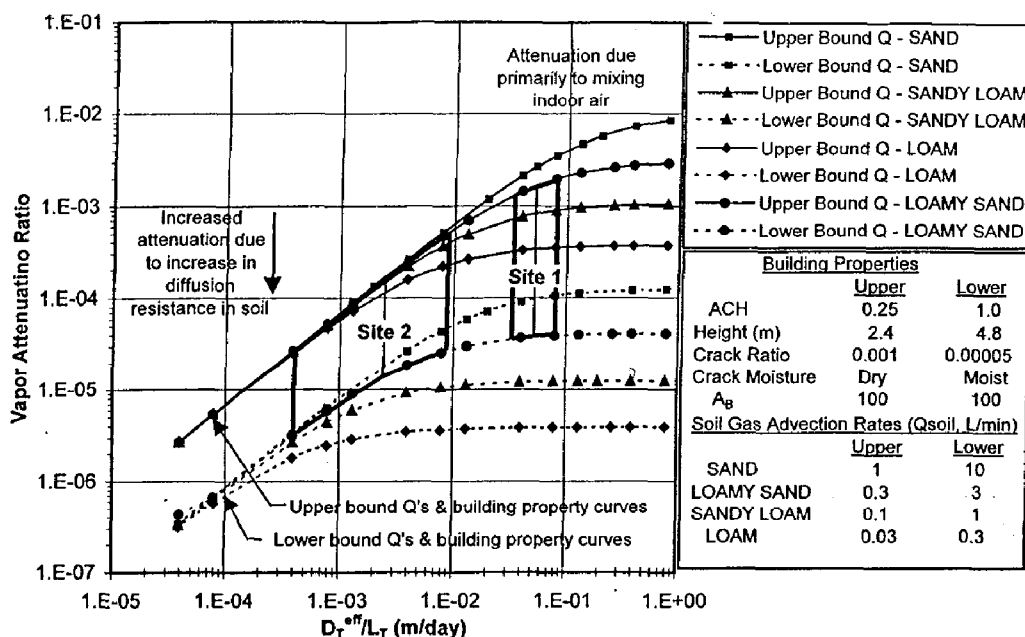


Figure 6. Predicted vapor attenuation ratio (benzene) for vapor concentrations at source and indoor air using Johnson and Ettinger (1991) model. Figure adapted from Johnson et al. (1998). Dry dust-filled cracks: Total porosity = 0.3; moist dust-filled cracks: water-filled porosity = 0.1, and total porosity = 0.3.

Table 3
Uncertainty Analysis for Normalized Effective Diffusion Coefficient

Parameters	Best Estimate Values		Uncertainty
	Site 1	Site 2	
Input Parameters			
Contamination	Above WT	Dissolved in Gdw	N/A
Contamination depth (m)	1.5	6.0	constant
U.S. SCS soil classification	Sandy Loam	Sandy Loam	N/A
Total porosity (θ)	0.390	0.390	+/- 10%
$S_R (\theta_w/\theta)$ above CZ (S)	0.265	0.265	+/- 25%
Height of CZ (L_{CZ}) (m)	N/A	0.250	+/- 25%
$S_R (\theta_w/\theta)$ in CZ (S_{CZ})	N/A	0.821	+12/-10%
Calculated Values			
D_T^{eff}/L_T lower est. (m/day)	0.0325	0.00038	
D_T^{eff}/L_T best est. (m/day)	0.0512	0.00248	
D_T^{eff}/L_T lowest est. (m/day)	0.0775	0.00861	
D_T^{eff}/L_T upper/lower range	2.4	23	

Notes: CZ = capillary zone, S_R = relative saturation, Gdw = Ground water, WT = water table.

predicted source vapor concentration (i.e., directly above the water table) estimated using the Henry's law constant assuming equilibrium partitioning between the dissolved and vapor phases. When measured source vapor concentrations are available, the α can be directly calculated. Because some deviation from equilibrium conditions would be expected, the α_m estimated using ground water and soil vapor data are not directly comparable. A key challenge for this approach is that there are numerous other "background" sources of VOCs in indoor and outdoor air for most chemicals of concern at contaminated

sites (Hers et al. 2001). The intrusion of soil vapor into buildings is also highly dependent on site-specific conditions and may vary over time. These factors complicate the interpretation of indoor air measurements when the goal is to deduce the subsurface-derived component.

The *tracer method* involves measurement of the indoor air concentration of a tracer injected below ground (SF_6), or a natural tracer such as radon (Fisher et al. 1996; Garbesi et al. 1993). The measured vapor intrusion for the tracer is, in turn, used to infer intrusion for the VOC of interest. Key factors affecting this approach are that boundary conditions for a tracer injected below a building may be different than those for the VOC of interest (e.g., if contamination is relatively deep) and that typically, an essentially inert tracer is used. When compared to the tracer, the mass loss or attenuation through sorption and/or biodegradation will be greater for most VOCs of interest. For these reasons, the tracer method will typically provide a conservative estimate of intrusion.

The *flux chamber* method involves measurement of soil-gas flow and/or VOC flux through cracks or openings in a building foundation. There are only a few published reports documenting the use of flux chambers to measure VOC flux into buildings (Figley and Snodgrass 1992; Hers and Zapf-Gilje 1998). Challenges for this approach are that these tests are difficult and costly to perform, and the uncertainty associated with "scaling up" the results for a small crack to an entire building.

Results and Discussion of Field Studies and Model Predictions

Indoor VOC Method

Vapor attenuation ratios are evaluated for 11 sites. The sites represent studies available to the authors with reasonable quality field data, and are for residential houses, ground-floor

Table 4
Measured and Model-Predicted Vapour Attenuation Ratios

Site & Reference	Contaminant or Tracer	Building and Foundation Type	Soil Conditions	Depth (m) ¹	Chemical	Source Concentration (ug/L)	N Indoor Air ²	α_m Stat	Measured α_m	J&E model α_p ³	Comments
Indoor VOC Method											
"Virginia (Motiva) Site", Fan and Quinn (2000)	petroleum HC, NAPL above water	SFR, basements, attached garages, cement block foundations	claystone saprolite k = 0.01 darcy	0.5	benzene	V: 410	13	50th	<8.3E-6	3.70E-06	
"Chatterton Site" Delta, B.C. Canada Hers et al. (1998) Hers et al. (2000a)	BTX	research greenhouse	surface silt to f. sand, underlain by m. sand with k = 10 darcies	1.4	benzene	V: 15,000	3-4	Avg	< 5.3E-7	1.3E-05	C1: $\Delta P = 0$ Pa, $\eta = 3.3E-4$
	petro-	slab-on-grade			toluene	V: 20,000	3-4	Avg	< 1.9E-6	1.3E-05	C1: $\Delta P = 0$ Pa, $\eta = 3.3E-4$
	chemical	poured concrete			benzene	V: 15,000	3-4	Avg	4.0E-07	5.9E-05	C2: $\Delta P = 2.5$ Pa, $\eta = 1E-4$
	plant,	2 mm edge crack			toluene	V: 20,000	3-4	Avg	5.9E-07	5.9E-05	C2: $\Delta P = 2.5$ Pa, $\eta = 1E-4$
	NAPL above water table				benzene	V: 15,000	3-4	Avg	9.9E-05	7.8E-05	C3: $\Delta P = 10$ Pa, $\eta = 1E-4$
					toluene	V: 20,000	3-4	Avg	1.3E-04	7.8E-05	C3: $\Delta P = 10$ Pa, $\eta = 1E-4$
					benzene	V: 15,000	3-4	Avg	7.2E-06	8.0E-05	C4: $\Delta P = 10$ Pa, $\eta = 3.3E-4$
					toluene	V: 20,000	3-4	Avg	3.4E-05	8.0E-05	C4: $\Delta P = 10$ Pa, $\eta = 3.3E-4$
					benzene	V: 15,000	3-4	Avg	5.8E-06	2.9E-05	C5: $\Delta P = 30$ Pa, $\eta = 3.3E-4$
"Paulsboro Site", NJ USA, Laubacher et al. (1997)	gasoline NAPL above water table	SFR basement	sand, some silt	2.74	benzene	V: 576	15	Avg	<1.6E-6	4.3E-04	
					iso-pentene	V: 28,000	1	N/A	<9E-7	2.46E-04	
"Alameda (Air Station) Site", CA, USA Fischer et al. (1996)	gasoline NAPL above water table	small commercial building, slab-on-grade poured concrete	sand k = 1 to 3 darcy	0.7	benzene	V: 200	1	N/A	<9E-6	2.45E-04	
"Mass. DEP Sites" USA, Fitzpatrick & Fitzgerald (1996)	petroleum hydrocarbon	N/A (3 sites)	N/A	N/A	benzene	N/A	-	N/A	1E-5 to 4E-5	INS	
"Midwest School Site" USA, Moseley and Meyer (1992)	petroleum HC NAPL above water table	Built 50's, at-grade construction, crawl-space, large paved area	sand & gravel, discontinuous clay lenses	~ 3	benzene total HC	N/A	N/A	N/A	HC-like odours -- 1E-4	INS	crawlspace conc.: benzene ~ 8.3 mg/m ³ , Total HC ~ 500 mg/m ³
"CDOT HDQ Site" Colorado, USA Johnson et al. (2000)	chlorinated solvents, dissolved plume	mostly apartments, few SFRs, mostly slab-on-grade, few crawl-spaces & basements, AC mostly window units, heating natural gas, baseboard, and/or fireplaces	weathered & fractured claystone above water table	4.6	1,1 DCE	G: 10-10,000	115-150	Geom	4.8E-06	2.0E-05	α_m values for houses above plume with DCE groundwater concentration > 10 ug/L
					1,1 DCE	G: 10-10,000	150	90th	2.0E-05		
					TCE	G: 3-3,000	115-150	Geom	1.4E-05	7.0E-05	
					1,1,1 TCA	G: 10-1,000	115-150	Geom	1.7E-05	6.6E-05	
					1,1,1 TCA	G: 10-1,000	150	90th	6.6E-05		
					above 3 CS		115-150	Geom	1.2E-05	8.6E-05	average for 3 chlorinated solvents (CS)
"Redfields Site" Colorado, USA Envirogroup (1999)	chlorinated solvents, dissolved plume	SFRs, built 50's and 60's, mostly basements or crawlspaces, no combustion air intakes	clay & silt, some sand layers, mostly sand or silt near WT	6.1 to 7.3	1,1 DCE	G: 10-1,000	65	50th	1.50E-05	INS	α_m values for houses above plume with DCE groundwater concentration > 10 ug/L
					1,1 DCE	G: 10-1,000	65	Avg	7.60E-05		
					1,1 DCE	G: 10-1,000	65	90th	1.20E-04		
Hamilton Site Colorado, USA (2001), unpublished	chlorinated solvents, dissolved plume	SFRs, built 50's mostly basements	primarily sand & gravel, some clay & silt layers	9.7 to 11	1,1 DCE	G: 15-30	32	50th	6.80E-05	INS	Gravel at water table
						G: 15-30	32	90th	1.40E-04		
"Lowry (Air Force Base) Site" Colorado, USA Versar (2000)	chlorinated solvents, dissolved	SFR: mostly basements some crawlspaces	silty sand to silt, generally silty sand near water table	6.1 to 7	1,1 DCE	G: 1.4-1.9	>50	50th	2.20E-05	INS	max $G_{ss} \alpha_m = 6.2E-04$
					TCE	G: 120-170	>50	50th	2.20E-05		max $G_{ss} \alpha_m = 1.2E-03$
					1,1 DCE	V: > 29	>50	50th	6.50E-04		max $V_{ss} \alpha_m = 8.3E-03$
					TCE	V: > 1,000	>50	50th	7.70E-04		max $V_{ss} \alpha_m = 1.4E-02$
"Mountain View Site" California, USA Wu (2000)	chlorinated solvents, leach-field & dissolved ⁵	SFRs, built 1998, at-grade construction with moisture vapor barrier	mostly silty/clayey sand & gravel, some sand or silt lenses	1.5	TCE	V: 84	14	Max	2.80E-04	INS	α_m shallow vapour
						V: 84	14	2nd ⁶	<1.3E-5		
					TCE	G: 735	14	Max	7.80E-05		α_m groundwater, depth to groundwater = 10.7 m
"Mass. DEP Sites" USA, Fitzpatrick & Fitzgerald (1996)	chlorinated solvents	N/A (19 sites)	N/A	N/A	CS	N/A	N/A	N/A	2E-6 to 1E-1	INS	high α_m associated with highly permeable building envelopes (earthen floor, block walls & sumps)
Tracer and Flux Chamber Tests											
"Central California Site", Garbesi & Sextro (1989)	SF ₆	SFR, basement poured slab, block walls coated with asphalt	sandy loam to loamy sand, k = 0.1 to 10 darcies	sub-slab	SF ₆	N/A	N/A	N/A	~ 1E-3	N/A	$\Delta P = 30$ Pa
"Alameda Site" Fischer et al. (1996)	SF ₆	small commercial, slab-on-grade, concrete	sand, k = 1 to 3 darcy	sub-slab	SF ₆	N/A	N/A	N/A	2E-4 to 4E-4	N/A	$\Delta P = 3$ (estimate based on wind loading)
U.S. Sites Little et al. (1992)	radon	SFRs	N/A	sub-slab	radon	N/A	N/A	N/A	1.6E-3 ⁷	N/A	
"Spokane River Valley Sites", WA, USA, Rezvan et al. (1992)	radon	SFRs (14), 8 houses slab-on-grade, 6 basement	highly permeable sand & gravel, k ~ 200 darcies	sub-slab	radon	N/A	N/A	N/A	~ 7.9E-3 to 4.5E-2	N/A	winter conditions, mean house volume = 500 m ³ , ACH = 0.5/hr

Notes: ¹Depth to contamination from underside of foundation slab; ²N = Number of indoor air samples tested; ³Best estimate unless otherwise noted; ⁴Upper range; ⁵Contamination likely in unsaturated zone; ⁶2nd highest α_m value; ⁷Alpha (α) estimated using mean radon content of soil combined with appropriate constant divided by radon concentration in U.S. homes (55 Bq m⁻³); N/A = not available or applicable, SFR = single family residence, SF₆ = sulphur hexafluoride; V = vapor, V_{ss} = sub-slab, G = ground water, bgs = below ground surface, HC = hydrocarbon, AC = air-conditioning, INS = insufficient data, ACH = air exchanges per hour, WT = water table, CS = chlorinated solvents.

Table 5
Input Parameter Values Used for Johnson and Ettinger (1991) Model¹

Parameter	Virginia Site	Chatterton Site	Paulsboro Site	Alameda Site	Midwest Site	CDOT HDQ Site	Redfields Site	Hamilton Site	Lowry Site	Mountain West Site
US SCS soil type used for D_T^{eff}/L_T	N/A	N/A	N/A	N/A	N/A	N/A	Loamy Sand	Sand	Loamy Sand	N/A
Depth to contamination (L_T) (m)	0.5	1.4	2.74	0.7	3.0	4.8	6.1	10.3	0.25 ¹	1.5 ²
Total porosity unsaturated zone (θ)	0.43	0.36	0.39	0.36	0.4	0.4	0.39	0.375	0.39	0.41
Air-filled θ unsaturated zone (θ_a)	0.28	0.21	0.23	0.22	0.25	0.26	0.287	0.319	0.287	0.2
Height of capillary zone (L_c) (m)	N/A	N/A	N/A	N/A	N/A	0.2	0.25	0.17	0.17	0.25
Total θ capillary zone (θ_c)	N/A	N/A	N/A	N/A	N/A	0.4	0.39	0.375	0.39	0.41
Air-filled θ capillary zone ($\theta_{a,c}$)	N/A	N/A	N/A	N/A	N/A	0.08	0.07	0.12	0.07	0.1
D_T^{eff}/L_T (m/day)	0.12 ⁴	0.023 ⁴	0.014 ⁴	0.054 ⁴	0.016 ⁴	3.4E-3 ⁶	2.4E-3 ⁷	8.4E-3 ⁷	0.49 ⁸	0.013 ¹⁰
Soil-air permeability k_a (10^{-12} m ²)	0.01	10	10	3	—	N/A ¹²	—	—	2.4E-3 ⁹	1.5E-3 ¹¹
Building underpressurization (Pa)	1	0, 2.5, 10, 30	5	3	—	N/A ¹²	—	—	—	—
Foundation crack ratio (η)	1.5E-03	3.3E-4 to 1E-4	1.E-04	1.E-04	—	1.E-04	—	—	—	—
x_{crack} (m)	55.9	26.8	27.6	26.8	—	N/A ¹²	—	—	—	—
z_{crack} (m)	2.0	0.3	2.13	0.2	—	N/A ¹²	—	—	—	—
Q_{soil} (L/min)	0.0016	8.2 to 29	2.8	2.2	—	10	—	—	—	—
Total θ dust-filled cracks (θ_{crack})	0.43	0.25	0.25	0.25	—	0.4	—	—	—	—
Air-filled θ dust-filled cracks ($\theta_{a,crack}$)	0.28	0.25	0.25	0.25	—	0.26	—	—	—	—
Air exchange per hour (ACH)	0.76	0.42 to 14.3	0.42	2.1	—	0.45	—	—	—	—
Building mixing height (m)	2.0	2.19	2.74	2.4	—	3.0	—	—	—	—
Subsurface building area (A_B) (m ²)	186	57	39	50	—	89	—	—	—	—

Notes: ¹Depth to sub-slab soil gas probes; ²Depth to shallow gas probes; ³Depth to ground water; ⁴Benzene; ⁵Iso-pentene; ⁶Average 1, 1 DCE, TCE and 1,1,1 TCA; ⁷1,1,DCE; ⁸DCE for sub-slab vapor source (TCE value is 0.43); ⁹DCE for ground water source (value for TCE is 2.2E-03); ¹⁰TCE for shallow vapor source; ¹¹TCE for ground water source; ¹² Q_{soil} is estimated directly; therefore x_{crack} , z_{crack} , ΔP and k_a not needed; ¹³Building foundation thickness not included since has negligible effect.

apartments, or small commercial buildings. Site characteristics and estimated input parameters are summarized, and measured and J&E model-predicted vapor attenuation ratios (α_m and α_p) are compared (Tables 4 and 5, Figure 7). In most cases, the vapor attenuation ratios are estimated by the authors using site data; in a few cases, the ratios given in references cited in Table 4 are reported. This has led to differences in the statistical estimators used to characterize the variability in α_m and α_p . For completeness, the vapor attenuation ratios reported for several Massachusetts sites are also included in Table 4; these sites are not included in the 11 case study sites discussed later.

The quality and quantity of site characterization data, and ability to distinguish measured indoor air concentrations from background VOC sources varies from site to site. For three sites, the VOC concentrations in a relatively large number of houses above the contaminant plume were significantly greater than house concentrations in background areas, resulting in fairly reliable α_m estimates. For the remaining sites, either the vapor-derived VOC concentrations in indoor air were significant in only a small subset of houses above the contaminant plume, or there was no significant difference between above plume and background indoor air concentrations. The vapor attenuation ratio is not measurable when there is no significant vapor-derived component; however, the indoor air concentrations can be used to calculate upper bound α_m values, represented as "less than" values in Table 4, and dashed lines in Figure 7.

For each site (except Chatterton), a predictive "envelope" for

α_p was generated. A best estimate D_T^{eff}/L_T was directly calculated when reasonably good quality moisture content data was available. When good quality data was not available, the U.S. SCS soil texture class was inferred based on soil descriptions and the simplified VG method was used to calculate D_T^{eff}/L_T . We recognize that inference of soil texture is approximate and subjective. The upper and lower bound D_T^{eff}/L_T values were approximated using the same variability calculated for the two hypothetical sites discussed earlier (Table 3). The upper and lower bounds for Q_{soil} and building properties are the curves presented in Figure 6. A Q_{soil} range of 1 to 10 L/min (i.e., representative of sand) was assumed for all sites (except Virginia) because either coarse soils were present below building foundations, or there was no information on soil type (in these cases, sand was assumed to be present below foundations). Based on the fine-grained near-foundation soils at the Virginia site, a Q_{soil} range of 0.03 to 0.3 L/min (i.e., representative of loam) was assumed. When there was sufficient information on building properties and soil gas advection potential, the J&E model-predicted α_p was also estimated (represented as symbols in Figure 7). For the Chatterton site, only the best estimate α_p were plotted because testing at this site involved an experimental building and test cases not representative of generalized predictive envelopes in Figure 6.

Measured Vapor Attenuation Ratios at Petroleum Hydrocarbon Sites

Case study sites with petroleum hydrocarbon contamination have coarse-grained soils (except for the Virginia site) and

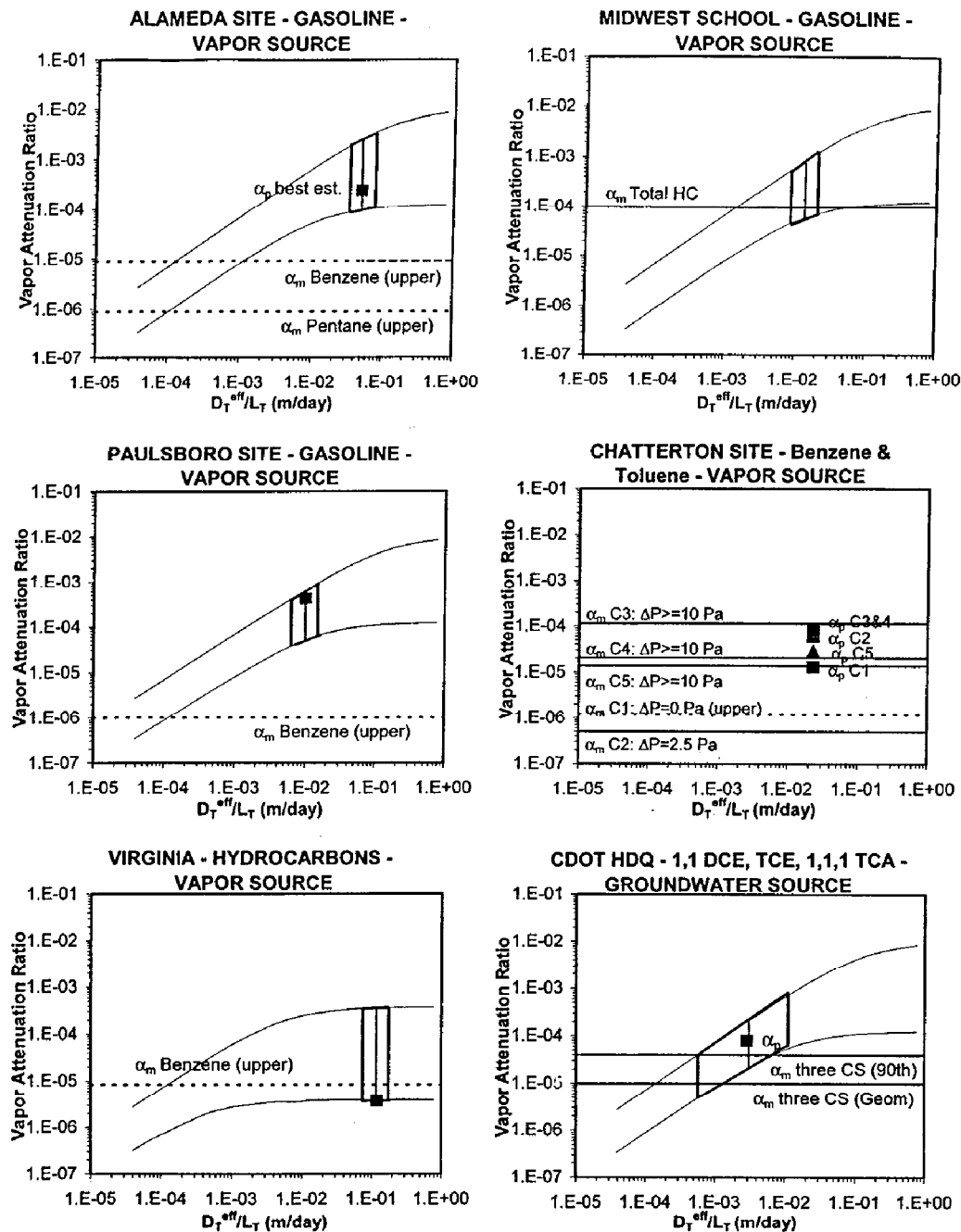


Figure 7a. Comparison between measured and J&E model-predicted vapor attenuation ratio (benzene). Upper and lower bound curves from Figure 6 are included. Dashed lines indicate that α_m is upper bound value. Symbols are best estimate α_p values.

shallow to moderate depths to contamination (0.5 to 3 m). Extensive residual nonaqueous phase liquid (NAPL) is present above the water table at the Chatterton site. There is evidence for some residual NAPL above the water table at the Alameda, Paulsboro, Virginia, and Midwest School sites. Indoor air testing was limited to a single or small number of buildings at each case study site. For petroleum sites, near-source vapor concentrations are available and therefore the α_m is directly calculated (vapor α_m).

At the Virginia, Chatterton (depressurization (ΔP) = 0 Pa case), Paulsboro, and Alameda sites, there was no difference between indoor air concentrations measured in building(s)

above the plume and in background areas, indicating that the α_m are unknown. For these sites, the α_m calculated using the measured indoor air concentrations are upper-bound values and range from $<4.0 \times 10^{-7}$ to $<9.0 \times 10^{-6}$. For the Chatterton $\Delta P = 2.5$ Pa case, there was a statistically significant difference in indoor and background indoor air concentrations; however, the α_m remained low (4.0×10^{-7} to 5.9×10^{-7}). For the Chatterton $\Delta P = 10$ and 30 Pa cases, there was a significant increase in indoor air concentrations and α_m .

At the Midwest School site, hydrocarbon-like odors were noted indoors during a period of relatively heavy rains and high water table in September 1992. Subsequent analysis of indoor

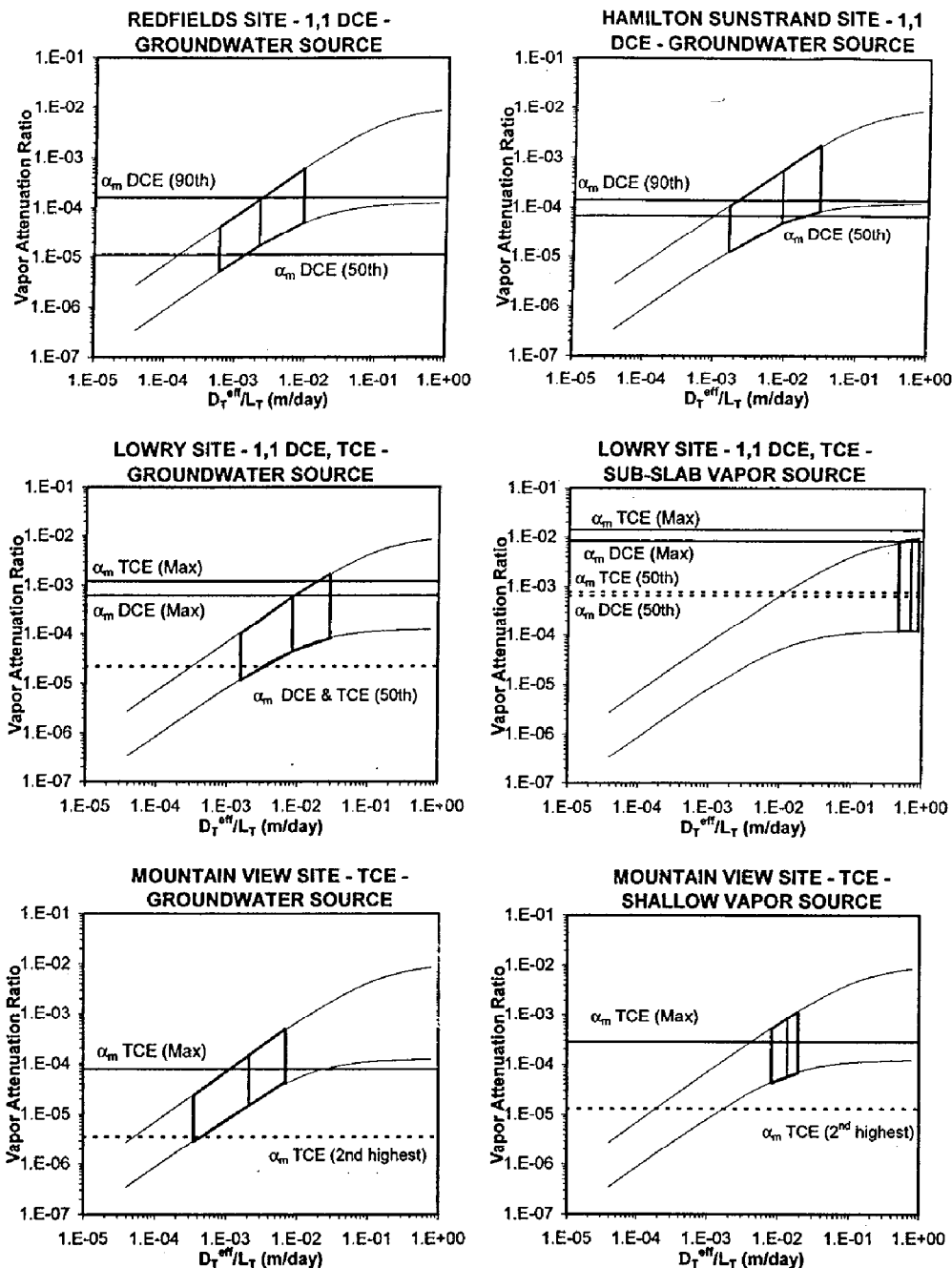


Figure 7b. Comparison between measured and J&E model-predicted vapor attenuation ratio (benzene). Upper and lower bound curves from Figure 6 are included. Dashed lines indicate that α_m is upper bound.

air during October 1992 indicated that hydrocarbon concentrations in indoor air were elevated but could not be conclusively distinguished from background sources at this time. However, the benzene (8 mg/m^3) and total hydrocarbon concentrations (500 mg/m^3) in an unventilated crawlspace below the ground floor were well above background levels. Based on a rough estimate of the source vapor concentrations and odor thresholds for hydrocarbons, the α_m may have been on the order of 1×10^{-4} .

Field data, including soil vapor profiles, indicate there was significant bioattenuation of hydrocarbon vapors for the Alameda and Chatterton ($\Delta P = 0$ and 2.5 Pa cases) sites. This

is consistent with other studies indicating that biodegradation can result in significant vadose zone attenuation of hydrocarbon vapors, provided sufficient O_2 is present (Ostendorf and Campbell 1991; Ririe and Sweeney 1995). For higher underpressurizations (10 and 30 Pa), at the Chatterton site hydrocarbon vapor concentrations were elevated because of increased vapor flux from deeper soil, and reduced travel times (Hers et al. 2002). The relatively high α_m at the Chatterton site are from the combined effect of shallow contamination, relatively permeable soils, and high building underpressurizations.

The Paulsboro and Midwest School sites had elevated

hydrocarbon vapor levels directly below the building slab. For the Midwest School site, we speculate that elevated indoor hydrocarbon concentrations may have been a result of limited biodegradation owing to a large building and paved area, which reduced oxygen recharge, combined with factors that contributed to vapor intrusion into the building. These factors include building construction (i.e., crawlspace) and/or a sanitary sewer that was located near the water table within the hydrocarbon plume, which may have acted as a preferential pathway. At the Virginia site, contamination was shallow but no significant vapor intrusion was measured possibly because of the presence of fine-grained soils and/or building construction (i.e., tight foundations).

Comparison to Model Predictions for Petroleum Hydrocarbon Sites

Comparisons for the Chatterton ($\Delta P = 0$ and 2.5 Pa cases), Paulsboro, and Alameda sites indicate that the best estimate α_p are one to two orders of magnitude higher than the measured or upper bound α_m indicating the J&E model results in conservative predictions for these sites. Comparisons for the Chatterton ($\Delta P = 10$ and 30 Pa cases) and Virginia sites indicate the best estimate α_p are similar to the α_m . The high soil-gas advection rates for the Chatterton site resulted in significant vapor intrusion rates and hence similar α_p and α_m . For the Virginia site, the α_p is lower than at other sites owing to the influence of the fine-grained soils. For the Midwest site, the predictive envelope for α_p also intersects the α_m ; however, the α_m is highly uncertain.

Measured Vapor Attenuation Ratios at Chlorinated Solvent Sites

At four case study sites with chlorinated solvent contamination (CDOT, Redfields, Hamilton, and Lowry), dissolved plumes have migrated below houses (Table 4). The depth to the water table at these sites ranged from ~4.8 to 10.7 m below ground surface. The ground water plumes at these sites are relatively long and narrow, resulting in significant spatial variability in dissolved ground water concentrations. At the fifth site (Mountain View), houses were constructed on top of a former leach field where chlorinated solvents had been disposed of. Therefore, in addition to ground water, shallow soil is likely contaminated at this site. Soil grain size at the sites is variable (Table 4). For all sites, the α_m are estimated using vapor concentrations predicted from ground water data (ground water α_m). For the Lowry and Mountain View sites, soil vapor data were also available; therefore, the α_m is also directly calculated using vapor data (unless otherwise noted, the α_m given below are for the ground water source scenario).

For the CDOT site, the differences in three chlorinated solvent concentrations (1,1 DCE, TCE, and 1,1,1 TCA) in houses above the plume and at background locations are statistically significant. However, the ground water and indoor air data were found to be unreliable at the periphery of the plume and therefore low ground water and indoor air concentrations were removed from the database prior to calculating the α_m . The resulting database comprises several hundred tests from apartments and houses. The methodology used to estimate α_m is further described in Johnson et al. (2000). The geometric mean and 90th percentile α_m for the CDOT site are 1.0×10^{-5} and 5.2×10^{-5} . Analysis of the intrusion database for the site indi-

cated no strong correlation between seasons and α_m , or difference between basement and slab-on-grade construction (personal communication, Dr. Jeff Kurtz, EMSI Inc.).

For the Redfields site, the difference in 1,1 DCE concentrations in houses above the plume and at background locations are statistically significant. A data screening procedure similar to that used for the CDOT site resulted in α_m only being estimated in areas where the 1,1 DCE concentrations in ground water exceeded 10 $\mu\text{g/L}$. A visual interpolation method was used to estimate ground water concentrations below houses. The resulting database comprises 65 houses nearest to the Redfields site. The 50th and 90th percentile α_m for the Redfields site are 1.1×10^{-5} and 1.2×10^{-4} . Synoptic data for the Redfields site indicated a slight correlation between indoor 1,1 DCE concentrations and season, for some houses, with winter-time values that were two to three times higher than summer-time values (personal communication, Dr. David Folkes 2000).

For the Hamilton site, the difference in 1,1 DCE concentrations in houses above the plume and at background locations are statistically significant. Because ground water data was limited, the attenuation ratio analysis is for a strip of 32 houses parallel and closest to the long axis of the plume (and wells) in the area with 1,1 DCE concentrations above ~10 $\mu\text{g/L}$. The 50th and 90th percentile α_m for the Hamilton site are 6.8×10^{-5} and 1.4×10^{-4} .

At the Lowry site, the database evaluated consists of more than a year of quarterly testing at 13 houses above and near the periphery of the plume. Concurrent testing of indoor air, and subsurface vapor concentrations for houses with slab-on-grade or basement construction, and crawlspace air for houses with crawlspaces was conducted. At one house, the maximum TCE and 1,1 DCE concentrations in indoor air were 51 $\mu\text{g/m}^3$ and 0.91 $\mu\text{g/m}^3$, suggesting significant vapor intrusion. At three other houses, the TCE concentrations in indoor air were mostly between 5 and 15 $\mu\text{g/m}^3$. Compared to published background data for TCE (Hers et al. 2001) and data for houses along the periphery of the plume, it is possible that concentrations at these three houses included a soil vapor-derived component. The indoor air concentrations were at background levels in remaining houses.

Measured vapor attenuation ratios are estimated for a subset of four Lowry houses with nearby ground water data. For this data subset, the maximum indoor air TCE concentration was 51 $\mu\text{g/m}^3$, but exceeded 5 $\mu\text{g/m}^3$ in only one house. Therefore, most α_m are upper bound values. When all data are used, the 50th percentile and maximum ground water α_m are 2.2×10^{-5} and 1.2×10^{-3} for TCE, and 2.2×10^{-5} and 6.2×10^{-4} for 1,1 DCE. The maximum, as opposed to 90th percentile α_m , was calculated owing to the relatively limited number of tests for this site. The Lowry subsurface vapor concentrations were highly variable and elevated below certain houses (e.g., TCE up to 10,000 $\mu\text{g/m}^3$), but near background levels below other houses above the plume. An analysis of the house data subset where indoor air TCE concentrations exceeded 5 $\mu\text{g/m}^3$ and/or subsurface TCE concentrations exceeded 1000 $\mu\text{g/m}^3$ indicated that the 50th percentile and maximum subsurface vapor α_m are 7.7×10^{-4} and 1.4×10^{-2} . Available synoptic data for the Lowry site indicated no significant seasonal variation in subsurface or indoor air concentrations.

At the Mountain View site, indoor air in seven houses

above the contaminated area and two "background" houses in a noncontaminated area was tested on two occasions. The indoor TCE concentration in one house was 12 and 25 $\mu\text{g}/\text{m}^3$, whereas the TCE concentrations in remaining houses were at background levels (0.26 to 1.1 $\mu\text{g}/\text{m}^3$) (Wu 2000). The maximum ground water α_m is 7.8×10^{-5} while the shallow vapor maximum α_m is 2.8×10^{-4} .

When all five sites are evaluated, the results can be summarized as follows. The 50th percentile (or geometric mean) and 90th percentile (or maximum) α_m values for the ground water to indoor air pathway were remarkably similar for all sites (approximately 1×10^{-5} and 1×10^{-4} , respectively). For individual sites, there is significant house-to-house variability in α_m (e.g., two order of magnitude difference for Redfields site); however, based on the available data there appear to be only slight, if any, seasonally induced variations in vapor intrusion, and similar intrusion rates for houses with basement and slab-on-grade construction. Potential sources of variability in α_m include inaccurate estimation of water table ground water concentrations below houses, geological heterogeneity, differences in house construction and depressurization, and differences in ventilation rates and house activities during indoor air testing. At the Lowry and Mountain View sites, no significant vapor intrusion could be measured for most houses. One likely reason for the generally nonsignificant intrusion is that ground water concentrations are lower at these sites, compared to the CDOT, Redfields, and Hamilton sites. Another possible factor for the Mountain View site is the building construction, which consists of at-grade foundation slab with (moisture) vapor barrier. Overall, the results suggest that geologic conditions and diffusion rates have the greatest influence on vapor intrusion rates at the chlorinated solvent sites, and that building factors are less important.

Comparison to Model Predictions for Chlorinated Solvent Sites

Comparisons for sites with the most reliable data (CDOT, Redfields, and Hamilton) indicates that the predictive envelope for the α_p intersects the α_m . The centroid of the predictive envelope is in all cases higher than the 50th percentile α_m suggesting, on average, the J&E model would result in conservative predictions. For the CDOT site, the best estimate α_p is approximately eight times higher than the 50th percentile α_m . For the Lowry site, the predictive envelope is below the α_m for one house with significant vapor intrusion, indicating a non-conservative prediction in this case. For the Mountain View site, the predictive envelope for α_p intersects the maximum α_m . Overall, the J&E model in most cases results in conservative predictions (i.e., α_p is higher than α_m). However, the comparisons highlight the potential for nonconservative predictions if a combination of low Q_{soil} and low D_T^{eff}/L_T are used.

Tracer Method

There are several sites where tracer tests can be used to estimate α , which range from $\sim 2 \times 10^{-4}$ at the Alameda site to 4.5×10^{-2} at the Spokane River (Valley) sites (Table 4). The Spokane River sites were calculated using an assumed average house volume (500 m^3) and building ventilation rate (air changes per hour (ACH) = 0.5 hour^{-1}) and therefore are approximate. Soils at the Spokane River site are very permeable, and α is based on winter conditions (i.e., highest expected

seasonal building depressurization); therefore, the α for this site is considered an upper range value. It should be remembered that tracer studies represent α values for near-field boundary conditions and, therefore, are not representative of intrusion at many sites contaminated with VOCs. The tracer test α values are, however, consistent with the upper range of the J&E model predictions (Figure 6).

Flux Chamber Method

A method that has been used for radon assessments is the equivalent leakage area (ELA) method (Grimsrud et al. 1982; CSG 1986). The ELA is obtained by developing an empirical relationship between the soil-gas flow into a building and building depressurization. Soil-gas flows are measured using flux chambers and mass flow meters. In one study involving multiple measurements of soil-gas flow through various building foundation cracks at 10 houses in Saskatchewan, Canada, the total house foundation ELA for the foundation edge cracks and utility penetrations ranged from 0.15 to 16.4 cm^2 (Figley and Snodgrass 1992). The contribution to total ELA from untrapped floor drains, present at a few houses, was excluded from this analysis since untrapped drains are uncommon in newer construction. For example, the National Building Code of Canada (1995) requires sealing of floor drainage systems that have the potential to allow soil-gas entry (Section 9.13.8.3).

The measured total ELA can be used to estimate soil-gas intrusion rates using the method in Figley (1997). A building depressurization representative of severe winter conditions (10 Pa), as proposed by Figley (1997), and possible values for the house volume (500 m^3) and building ventilation rate (0.3 ACH) produces α values between 3.6×10^{-4} and 3.8×10^{-2} . The α obtained in this manner is conservative because it assumes an unlimited and uniform soil-vapor source directly below the foundation slab (i.e., contaminants in vapor are replenished as fast as they are swept into the building).

Flux chamber tests have also been used to measure VOC flux rates through concrete cracks (Schmidt and Zdeb 1997; Hers and Zapf-Gilje 1998). Both studies indicated detectable VOCs were measured in soil gas transmitted through cracks, and the study by Hers and Zapf-Gilje (1998) indicated that the scaled-up flux for the entire building was of the same order as flux measured by the indoor VOC method.

Regulatory Implications

The J&E model is widely used for regulatory and guidance purposes in North America. Several agencies have developed generic screening criteria for the vapor intrusion pathway (Massachusetts 1993; Michigan 1998; Connecticut 1998). Semigeneric soil standards have been developed in Canada, based on two soil types (fine- and coarse-grained) and two building types (CCME 2000). Guidance recently developed by the U.S. EPA consists of a multitiered framework to evaluate the soil vapor intrusion pathway (U.S. EPA 2002). A primary (initial) screening step is used to identify sites with significant potential for vapor intrusion (e.g., odors, product in sumps or directly below foundation), and where indoor air monitoring and/or engineering controls is warranted. A secondary screening step involves the use of semigeneric curves for α , based

on soil type and depth, and target breathing concentrations in indoor air to back-calculate acceptable source ground water and soil vapor concentrations. Depending on the results of the secondary screening, there is the option to conduct a site-specific pathway assessment.

Derivation of regulatory criteria requires the prediction of cross-media transfer of contaminants, and vapor transport and intrusion into buildings. For the regulatory agencies cited previously (excluding Massachusetts), cross-media transfer between VOCs in ground water and soil vapor is predicted using the Henry's law constant assuming equilibrium partitioning. Under the Massachusetts guidance, the Henry's law constant is divided by 10 to account for source vapor concentrations that are typically lower than those predicted assuming equilibrium partitioning. The vapor attenuation ratios incorporated into regulatory criteria depend on whether the assumed contamination scenario is a dissolved ground water plume or an unsaturated zone contamination source. For a ground water source, the α incorporates vapor transport through both the capillary transition zone and unsaturated zone. For an unsaturated zone source, the α incorporates transport through just the unsaturated zone. For the agencies cited previously, the ground water source α ranges from 4.6×10^{-6} to 1.5×10^{-3} whereas the vapor source α ranges from 3.9×10^{-7} to 6.2×10^{-3} . An analysis of the previous regulatory criteria indicates that the key factor affecting the α is the Q_{soil} value chosen or estimated for predictive purposes. Of lesser importance is the assumed generic or semigeneric soil type.

When vapor attenuation ratios incorporated in regulatory criteria are compared to measured ratios for field studies presented in this paper, it is apparent that the low end of the regulatory range may not be conservative for some sites. Of greatest concern would be sites with nonbiodegradable chemicals, shallow to moderate depth contamination, and high advection potential (i.e., coarse soil, high building underpressurization).

Conclusions and Recommendations

A comprehensive evaluation of the J&E model characteristics and sensitivity, and comparisons of measured to model-predicted vapor attenuation ratios (α_m and α_p), have been provided for residential houses, ground-floor apartments, and small commercial buildings. Based on this analysis, the following conclusions can be drawn:

1. The J&E model is moderately too highly sensitive to soil-gas advection rate into the building (Q_{soil}), at D_T^{eff}/L_T values above $\sim 1 \times 10^{-3}$. Except when Q_{soil} is low, the J&E model is relatively insensitive to building foundation properties. At best, the range or uncertainty in J&E model predictions is about one order of magnitude when relatively good quality site-specific data is available.
2. Estimation of effective diffusion coefficient is subject to considerable uncertainty. Some of this uncertainty can be reduced through better site characterization, including careful lithological descriptions, testing of moisture content, grain size distribution and water retention, and appropriate consideration of the effect of surface barriers on soil moisture content.
3. Several radon and VOC tracer studies indicate that measured Q_{soil} values at coarse-grained soil sites, for single fam-

ily residences, ranged from ~ 1 to 10 L/min. Depending on the input values chosen, much lower Q_{soil} values can be predicted using the soil-gas advection model typically used in conjunction with the J&E model.

4. There are only a limited number of high quality and comprehensive field studies that can be used to help validate models for the vapor intrusion pathway.
5. For petroleum hydrocarbon sites, the vapor α_m for the Chatterton site (high ΔP cases) and Midwest site were on the order of 1×10^{-5} to 1×10^{-4} (the Midwest value is uncertain). For the remaining cases and sites, the possible upper bound vapor α_m ranged from $\sim 5 \times 10^{-7}$ to 1×10^{-5} .
6. For chlorinated solvent sites, the ground water α_m were on the order of 1×10^{-6} to 1×10^{-4} for the three sites with the most reliable data sets (CDOT, Redfields, and Hamilton). For one site with a smaller and somewhat less reliable data set (Lowry), the maximum ground water α_m was $\sim 1 \times 10^{-3}$ while the maximum subslab vapor α_m was $\sim 1 \times 10^{-2}$.
7. For the tracer and flux chamber studies, the α_m was on the order of 1×10^{-4} to 1×10^{-2} . In the context of VOC intrusion, these α_m represent conservative upper bounds owing to boundary conditions and tracer properties that are generally different than those at VOC-contaminated sites.
8. For almost all case studies, the best estimate J&E model-predicted α_p were one to two orders of magnitude less than the 50th percentile or median α_m , indicating that when best estimate and average conditions are evaluated, the J&E model predictions are conservative. There were a few cases studies where the best estimate α_p was less than the 90th percentile or maximum α_m , indicating the J&E model predictions are nonconservative for a small subset of houses or apartments. The comparisons also highlight the potential for non-conservative model predictions if a combination of low Q_{soil} and low D_T^{eff}/L_T are used.

The observed variability in α_m between different field sites, and individual houses at some sites, highlights the complexity of processes affecting vapor intrusion. Numerous factors potentially affect the vapor intrusion pathway including biodegradation, chemical transformation, sorption, contaminant source depletion, geologic heterogeneity, soil properties (moisture content, permeability, organic carbon content), building properties, meteorological conditions, and building ventilation rates. In light of this complexity, it is important to recognize the vapor intrusion modeling paradigm typically followed is a compartmental model for steady-state one-dimensional diffusion through soil, and diffusion and advection through a building foundation having an idealized edge or perimeter crack (J&E model). Often, a homogeneous soil is assumed, although it is relatively easy to model diffusion for multiple soil layers assuming site information is available (Johnson et al. 1998). Simulation of vapor transport through the building foundation and mixing of VOCs within the building airspace is highly simplified. Although not used for this study, it is noted that the J&E model has been modified to include first-order biodegradation for a dominant soil layer (Johnson et al. 1998) and oxygen-limited first-order biodegradation (Johnson et al. 2001).

Notwithstanding the above, the question remains: Can the

J&E model (or other similar screening models) be reliably used for the vapor intrusion pathway? Our answer is a qualified yes, provided that appropriate input values are used and the model sensitivity, uncertainty, and limitations are recognized. The answer may also depend on what the model is used for. For example, the use of the J&E model to set generic criteria is problematic owing to model sensitivity and uncertainty, and the wide range in possible site conditions. In our opinion, a semigeneric approach that incorporates site-specific information on critical factors affecting vapor intrusion (e.g., Q_{soil} and soil properties) improves on a single criteria approach. The technically preferred approach is to use the J&E model on a fully site-specific basis, and to calibrate model predictions using soil vapor profiles, and when possible, indoor air data. In all cases, an appropriate framework for model use and understanding of model characteristics is essential when using models for regulatory purposes.

Several data gaps and sources of uncertainty remain. Additional field-based studies should be conducted to evaluate the vapor intrusion pathway for different site conditions, and to more fully assess specific factors affecting vapor intrusion. Data that would contribute to a more in-depth pathway analysis include soil properties such as moisture content and porosity, soil vapor concentration profiles below buildings, building properties such as depressurization, and meteorological data. Further evaluation of biodegradation kinetics for hydrocarbon vapors, effect of surface barriers (e.g., buildings) on biodegradation, and chlorinated solvent transformation processes are also needed.

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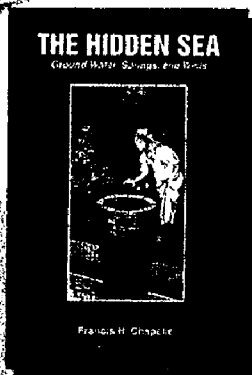
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